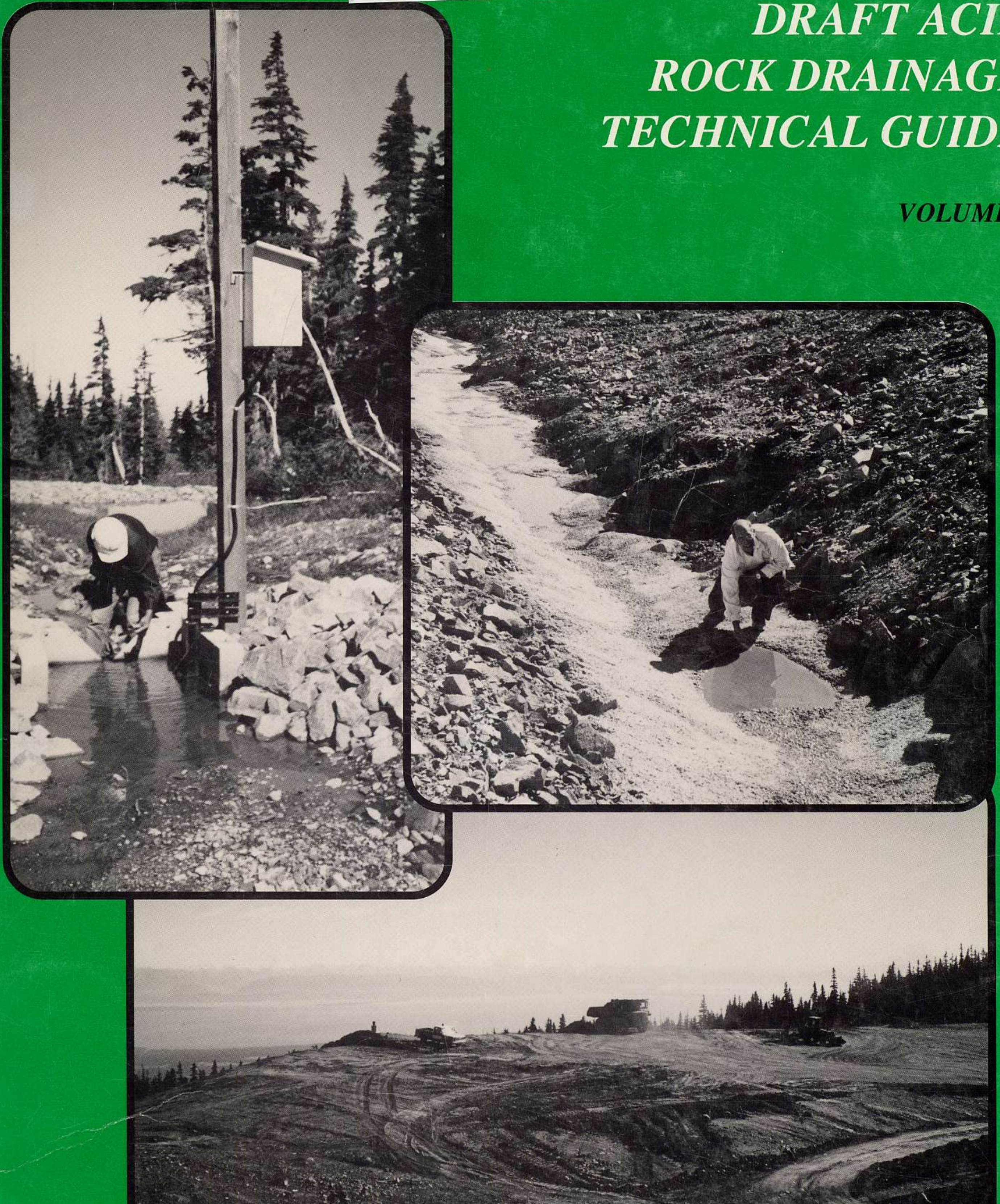
DRAFT ACID ROCK DRAINAGE TECHNICAL GUIDE

VOLUME 1



DRAFT ACID ROCK DRAINAGE TECHNICAL GUIDE

VOLUME 1

BRITISH COLUMBIA ACID MINE DRAINAGE TASK FORCE REPORT

Prepared for the:

British Columbia Acid Mine Drainage Task Force, with funding from Energy, Mines and Resources Canada and the British Columbia Ministry of Energy, Mines and Petroleum Resources under the Canada/British Columbia Mineral Development Agreement (1985-90).

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DISCLAIMER

The purpose of this guide is to provide the user with an understanding of the processes of acid generation and metal leaching from sulphide wastes and to provide guidance and recommendations in the application of currently available technology. This guide is issued as a draft document with the intention of being updated as technology progresses. The user of this guide assumes full responsibility for the design of facilities and for any action taken as a result of the recommendations contained in this guide. Neither the Province of British Columbia Acid Mine Drainage Task Force nor the consultants who prepared this document may be held liable for the outcome of any action taken by the user.

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	6.7 & 6.8	Norecol Environmental Consultants
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	6.10.3 & 6.10.4	Norecol Environmental Consultants
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CHAPTER 1: INTRODUCTION TO ACID ROCK DRAINAGE TECHNOLOGY

1.1 OBJECTIVES OF THIS TECHNICAL GUIDE

The objectives of this Guide are to provide the user with an understanding of the process of acid rock drainage (ARD) and to provide guidance and recommendations in the application of state-of-the-art technology in prediction, control and monitoring of ARD. The document is intended to be a working reference to government, industry, academicians and the interested public. The conclusions and recommendations contained in the Guide have been made for the benefit of the metal mining industry in British Columbia. Although reference is often made to ARD and related research within the coal mining industry world-wide, the recommendations may not necessarily apply to coal mining. It is intended that the Guide be updated to include developments made in research and practical field experience as these occur. The guide is intended to provide assistance to both operators and proponents of mining projects in the requirements of ARD prediction, control and monitoring, how these should be approached, and how available options should be evaluated. The recommendations contained in the guide are presented with reference to the requirements of the Province of British Columbia Mine Development Review Process.

The Guide consists of two volumes; this document, Volume I, contains detailed technical information while Volume II is a condensed "stand-alone" summary of recommendations for prediction, control and monitoring of ARD. Volumes I and II have been prepared for the Province of British Columbia Acid Mine Drainage Task Force by Steffen Robertson and Kirstern (B.C.) Inc. in association with Norecol Environmental Consultants and Gormely Process Engineering.

1.2 DESCRIPTION OF ACID ROCK DRAINAGE

Acid rock drainage (ARD) is the term used in this Guide to define drainage that occurs as a result of natural oxidation of sulphide minerals contained in rock which is exposed to air and water. This phenomenon is often referred to as acid mine drainage (AMD), however it is not necessarily confined to mining activities but can occur wherever sulphide-bearing rock is exposed to air and water. Some natural springs are acidic, usually in the vicinity of outcrops of sulphide-bearing rock. For practical purposes, the principal ingredients in the ARD process are; reactive sulphide minerals, oxygen and water. The oxidation reactions are often accelerated by biological activity. The chemical and biological reactions yield low pH water which has the potential to mobilize any heavy metals that may be contained in the waste rock or elsewhere. If water is available as a transport medium, the resultant drainage can contain products of the acid generation process, typically elevated metal levels and sulphate. This drainage can cause a detrimental impact on water quality in the receiving environment.

The process of mining sometimes results in the exposure of mine wastes, tailings or mine workings which contain sulphides in sufficient quantities to result in acid generation. Not all operations that expose sulphide-bearing rock will result in acid rock drainage. Acid drainage will not occur if either the sulphide minerals are non-reactive or if the rock contains sufficient alkaline material to neutralize the acid. If alkaline material is available in the rock waste, the pH may be raised as a result of neutralizing reactions as the drainage passes through the waste. The drainage that emerges from the waste has a pH that can vary from less than 1 to greater than 7. The quality and rate of release of ARD is governed by various chemical and biological reactions at the source of acid generation and along the drainage path from the source to the environment. If ARD is not controlled it can pose a very serious long-term threat to the environment due to the toxicity of heavy metals and other products to, for example, aquatic life.

A waste rock pile is shown schematically in Figure 1.2-1, as an example, illustrating the general process of acid generation and migration. Figure 1.2-1 shows a mixture of sulphidic and basic rock in the pile, the potential sources of oxygen and water, and acid generation/neutralization occurring where these ingredients are in contact. The illustration shows water percolating through the dump, coming into contact with both acid generating and neutralizing materials, before emerging from the toe as acid rock drainage.

1.3 BRIEF HISTORY OF ACID ROCK DRAINAGE

Acid rock drainage has likely been experienced from mines since mining started. Paine (1987) points out that a document summarizing available mining and smelting technology, written by Georgius Agricola and published in 1556, probably contains the first reference to environmental impacts of ARD. In countries such as Norway, which have been msjor miners and producers of copper and sulphide concentrates for the industrialized world for over 300 years, ARD has occurred extensively, continuously and in close association with human habitation for centuries (NIVA, 1987). The effect of ARD on man himself is limited but the impact on the aquatic environment can be large. In Norway, little control has been exercised in the prevention of ARD. From a total of thirty-one metal mines there has been minor environmental effects at fourteen, significant effects at three, and severe effects (resulting in the loss of a number of tens of kilometres of fish habitat in receiving waters) at three. In Australia, ARD from waste rock at an abandoned uranium mine was principally responsible for the loss of aquatic fauna for some nine kilometres of the Finniss River in the Northern Territory (NTDME, 1987).

The coal mining industry in the United States of America has for decades been identified as a source of ARD. This is particularly the case for the Appalachian States in eastern USA. Although acid rock drainage had been observed as early as 1698 in Pennsylvania, the environmental impacts in this and other coal producing states during the early years of development were negligible. Consequently, during this early period

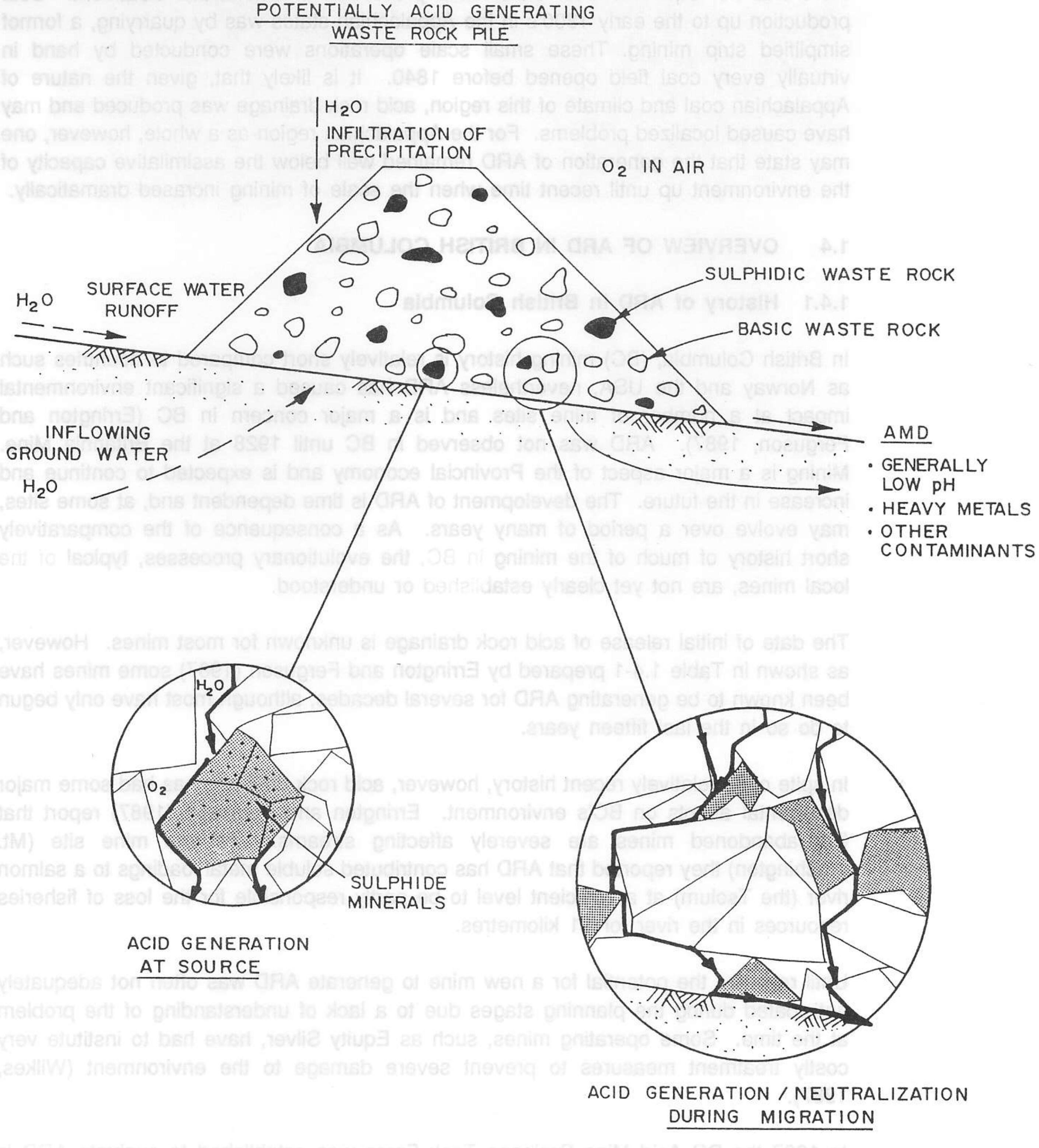


FIGURE 1.2-1
SCHEMATIC SHOWING CONCEPT OF ACID GENERATION AND ARD MIGRATION

there was no requirement for either ARD prevention or wastewater treatment. Coal production up to the early 1800's in the Appalachian states was by quarrying, a formof simplified strip mining. These small scale operations were conducted by hand in virtually every coal field opened before 1840. It is likely that, given the nature of Appalachian coal and climate of this region, acid rock drainage was produced and may have caused localized problems. For the Appalachian region as a whole, however, one may state that the generation of ARD remained well below the assimilative capacity of the environment up until recent time when the scale of mining incrased dramatically.

1.4 OVERVIEW OF ARD IN BRITISH COLUMBIA

1.4.1 History of ARD in British Columbia

In British Columbia, (BC) mining history is relatively short compared to countries such as Norway and the USA, nevertheless ARD has caused a significant environmental impact at a number of mine sites and is a major concern in BC (Errington and Ferguson, 1987). ARD was not observed in BC until 1928 at the Britannia Mine. Mining is a major aspect of the Provincial economy and is expected to continue and increase in the future. The development of ARD is time dependent and, at some sites, may evolve over a period of many years. As a consequence of the comparatively short history of much of the mining in BC, the evolutionary processes, typical of the local mines, are not yet clearly established or understood.

The date of initial release of acid rock drainage is unknown for most mines. However, as shown in Table 1.4-1 prepared by Errington and Ferguson (1987) some mines have been known to be generating ARD for several decades, although most have only begun to do so in the last fifteen years.

In spite of its relatively recent history, however, acid rock drainage has had some major detrimental affects on BC's environment. Errington and Ferguson (1987) report that five abandoned mines are severely affecting streams. At one mine site (Mt. Washington) they reported that ARD has contributed soluble metal loadings to a salmon river (the Tsolum) at a sufficient level to be partly responsible for the loss of fisheries resources in the river for 11 kilometres.

Until recently, the potential for a new mine to generate ARD was often not adequately anticipated during the planning stages due to a lack of understanding of the problem at the time. Some operating mines, such as Equity Silver, have had to institute very costly treatment measures to prevent severe damage to the environment (Wilkes, 1987).

In 1987 the BC Acid Mine Drainage Task Force was established to evaluate ARD in BC and to co-ordinate efforts for development of appropriate prediction and control technology. The Task Force consists of a number of committees with representatives from government, the Mining Association of British Columbia and academic bodies. In September 1987 the Task Force distributed a two-part questionnaire on ARD to all

TABLE 1.4-1

LAG TO FORMATION AND DURATION OF SOME ACID ROCK DRAINAGE IN BRITISH COLUMBIA

(After Errington and Ferguson, 1987)

Mine	Acid Mine Drainage	Mine Start Up	Mine Abandonment	Acid Mine Drainage Generated	Lag to Formation (Years)	Duration of Generation (Years)
Britannia	minewater	1905	1974	<1928	<23	>61
Duthie	tailings seepage	1952	1954	1980	<26	>9
Westmin	waste rock runoff	1966	operating	<1979	<13	>10
Equity	waste rock runoff	1980	operating	1981	1	8
Gibraltar	waste rock runoff	1971	operating	1982	11	7
Mt. Washington	waste rock runoff	1964	1966	<1972	<6	>17
Sullivan	minewater	1909	operating	<1950	<41	>39
Anyox	minewater	1914	1936	<1972	<36	>17

mining companies in the province. The objective of the questionnaire was to determine at which mines ARD is occurring and what measures are being implemented for the prediction, control, and monitoring of ARD. The results of the questionnaire are contained in a report issued in June 1988 (Steffen Robertson and Kirsten, 1988). Funding for these Task Force projects was obtained from government and industry.

1.4.2 ARD in British Columbia Today

1.4.2.1 COAL MINING

There are eight operating coal mines in British Columbia, none of which are presently generating acid rock drainage. All of these mines are located in the Rocky Mountain regions (south east and north east regions). Although the Rocky Mountain coal deposits appear to be non acid-generating, proposed coal mines in central BC and on Vancouver Island appear likely to encounter potentially acid generating strata. While a geographical trend in acid generation potential from coal mines may be suggested, insufficient information is available from mines in those areas where ARD is a potential problem to draw definite conclusions.

The coal deposits of the Rocky Mountain regions have relatively low sulphur contents (<0.6%). Proposed coal mines in central BC and Vancouver Island have encountered sulphur levels ranging from 0.6% to 4.2% (Errington and Ferguson, 1987), raising concerns for possible acid generation at these mines. Low sulphur contents in themselves are not necessarily sufficient to ensure that acid generation will not occur. One proposed mine in central BC has experienced low pH leachate from a coal waste pile, according to its response to the AMD Task Force questionnaire, even though it reports a sulphur content of only 0.4% (Steffen Robertson and Kirsten, 1988).

1.4.2.2 METAL MINING

Responses to the BC AMD Task Force questionnaire were received from sixteen operating metal mines in British Columbia, of which six generate acid rock drainage. Information was received from nine proposed metal mining operations and data was compiled for twenty-one mines that are either shut-down or abandoned. At least five abandoned mines are also known to generate ARD. In addition many new properties under active exploration have the potential for acid generation. ARD is controlled by various methods at the operating mines and abatement measures are being investigated for the abandoned mines. (Errington and Ferguson, 1987; Steffen Robertson and Kirsten, 1988). Acid rock drainage flows from underground workings, tailings, open pits and waste rock dumps. Acid producing mines are spread throughout widely separated regions of the province and include a variety of geologic environments. Table 1.4-2 summarizes features of the operating and abandoned mines in BC where ARD problems exist (Steffen Robertson and Kirsten, 1988).

TABLE 1.4-2

DETAILS OF MINES IN BC WITH ARD POTENTIAL

(After Steffen Robertson and Kirsten, 1988)

Type of Mine	Status	General Geology	% Sulphur in Waste	Type of Waste Facility Producing ARD
U and OP	O	MS	10 to 34% 4 to 8%	underground mine, open pit waste dumps
U	IS	MS	±30%	underground mine water
U and OP	A	MS	1 to 10%	underground mine, mine water tailings
OP	O	P	2%	waste rock
OP	O	P	N/A	tailings test plot, waste rock
OP	O	P	0 to 4.5%	waste rock
U and OP	O	MS	5 to 34%	tailings, underground workings
U	IS	PM	5.9%	tailings
U	A	MS	16%	underground mine water
OP	A	PM (Epithermal)	0.06 to 1.21%	waste rock, open pit
OP	O	MS	4 to 5%	waste rock

KEY:

Type of Mine	**General Geology**	**Mine Status**
U - Underground	MS - Massive Sulphide	O - Operating
OP - Open Pit	P - Porphyry	IS - Indeterminate shutdown
	PM - Precious Metal	A - Abandoned

1.5 LESSONS FROM OTHER PROVINCES AND COUNTRIES

The environmental impact of ARD is very evident in Norway where mining has been in progress for centuries with little or no control of ARD. The damage to the aquatic environment is extensive and many tens of kilometers of rivers and streams have been rendered barren. Considerable work is currently being carried out on evaluating abatement options. Control measures have or are being implemented at certain selected sites and these are to be closety monitored. One site has been offered to the Mine Environment Neutral Drainage (MEND) programme in Canada as a research project to evaluate the effectiveness of ARD control options. Considerable work has been carried out in Sweden on cover design technology which could be applied to BC conditions (Collin, 1987).

The effects of ARD and the results of abatement in the form of soil covers is demonstrated by the work carried out at the Rum Jungle site in Australia (NTDME, 1987). This is an abandoned uranium mine where a compacted clay cover has been placed over acid generating waste rock dumps. Initial monitoring indicates significant improvement in contaminant generation and transport from the waste while ongoing monitoring is necessary to determine the improvement in receiving water quality with time (Bennett et al, 1988). Important lessons have been learned from Rum Jungle in cover construction, maintenance, and monitoring (NTDME, 1987). ARD is a severe potential environmental problem in Australia, particularly in mining areas located in high precipitation regions. Nevertheless, it would appear that besides the work at Rum Jungle, little is currently being done in terms of research in this field.

Considerable ARD research has been conducted within the coal mining industry of the USA. Guidelines for ARD control, aimed primarily at surface mine operators in the Appalachian coal region in eastern USA, have recently been published (Skousen, et al, 1987). The principles and recommendations contained in the guidelines may be applied to coal and metal mining in BC although care should be exercised in applying experience from coal mining to hard rock metal mining. There are usually significant differences in chemical and particularly physical properties of the wastes from coal and metal mines. These factors may affect the evolution of ARD and may require different prediction, control and monitoring techniques.

Coal mining wastes in Eastern Canada are a major source of acid rock drainage as they are in the United States. It is considered worthwhile mentioning the location of acid rock drainage problems in the coal mining industry to illustrate the geographical variation in mineralization. Coal is produced in the Atlantic Region (New Brunswick and Nova Scotia) and Western Canada (Saskatchewan, Alberta and British Columbia). Atlantic coals are high pyritic sulphur coals, averaging 6- 10% total sulphur for the Minto Coals of New Brunswick and up to 4% total sulphur for the Sydney coals of Nova Scotia. Consequently, generation of acid rock drainage in the Minto and Sydney coal fields is more of a problem. By comparison, Western Canada coals are generally low sulphur coals, with typical total sulphur content of 0.37% (Errington and Ferguson, 1987). Acid rock drainage, due to pyritic slate being exposed during construction of Halifax International Airport in Nova Scotia, is reported to have caused an impact on surface water quality (Guilcher, 1987).

ARD from waste rock facilities at hard rock metal mines occurs predominantly in Western Canada. ARD also occurs from underground working and tailings, however, ARD from open pits and waste rock dumps is the major concern. On the other hand, in Eastern Canada, ARD from tailings is of major concern particularly at the underground copper and nickel mines in Ontario and Quebec. This trend is probably due to a number of factors. In BC metal mining in sulphide ores tends to be open pit mining with resultant large quantities of waste rock. The waste rock, while not necessarily high sulphur content, has low buffering capacity and is durable and hence permeable, allowing acid generation to occur and be realized rapidly. In Eastern Canada, underground mining is more predominant with lesser quantities of waste rock. The waste rock that is produced tends to be slaking which results in less permeable waste rock dumps.

1.6 ACID ROCK DRAINAGE RESEARCH IN CANADA

In BC the AMD Task Force is co-ordinating reseach and development of appropriate technology. A number of research projects in the areas of prediction and prevention, treatment and control, and monitoring are underway. The projects completed by 1989 and those scheduled for 1989 are described by Robertson (1989).

In response to the collective need to develop appropriate technology for ARD control, the Reactive Acid Tailings Stabilization (RATS) programme was initiated in 1981/82 with representatives from the Canadian mining industry, federal and provincial government. The programme, now re-named the Mine Environment Neutral Drainage (MEND) programme, has prepared a research plan to be undertaken over five years at a cost of $12,500,000. The objectives of this research plan are as follows:

- to provide a comprehensive scientific, technical and economical basis for the mining industry and governmental agencies to predict, with confidence, the long-term management requirements for reactive tailings and waste rock;

- to establish techniques that will enable the operation and abandonment of acid-generating tailings and waste rock disposal areas in a predictable, affordable, timely and environmentally acceptable manner.

It is the intention of the BC AMD Task Force to include such results and the results of international research in subsequent updates of the Guide as this data becomes available.

1.7 SOURCES OF ACID ROCK DRAINAGE

Acid rock drainage may occur from natural sources as well as locations where sulphidic rock has been exposed as a result of civil engineering, mining, or other activities. The sources of ARD from mining operations include:

- Drainage from underground workings.
- Surface runoff from open pit mine faces and pit workings.
- Waste rock dumps from metal mines and spoil piles from coal mining.
- Tailings deposits.
- Ore stockpiles and spent ore piles from heap-leach operations.

1.7.1 Underground Mines

The ARD from underground workings received early recognition since it generally occurs as point discharges of substantial flows of low pH water. Many of the earliest sources were drainage tunnels or mine adits. Examples of ARD from underground mines in British Columbia include the abandoned Britannia Mine, Anyox Project, and Tulsequah Chief Project (Steffen Robertson and Kirsten, 1988).

1.7.2 Open Pit Mines

The effects and consequences of ARD from open pit mines has become a cause of concern much more recently. This is because of the more recent history of large open pit mining in North America and the fact that many of the ARD producers or potential producers are still being operated or maintained, with ARD being used or treated. Many of these sites have been developed without plans to deal with ARD. The large areas of exposed rock inherent in open pits can result in large volumes of ARD. Long term slope deterioration can result in a continual exposure of fresh rock to the natural elements and hence ARD generation. An example of ARD from an open pit in BC is found at the abandoned Mt. Washington Mine on Vancouver Island (Steffen Robertson and Kirsten, 1988).

1.7.3 Waste Rock

Waste rock is generated by the excavation and construction operations undertaken to gain access to the ore body being mined, particularly for open pit operations. These wastes are exposed to precipitation, runoff and possibly seepage. Rock wastes which contain sulphides are a large source of ARD. The quantities of rock waste from earlier underground mining activities are proportionately less than the more recent large open pit mines. Thus the potential loads of ARD to surface waters has increased significantly with the more recent open pit mining operations. The chemical and physical properties of the waste rock dump will significantly affect the quality of ARD and the change in quality with time. Waste rock with potential to produce ARD occurs at the Equity Silver Mine, Bell Mine, and Lynx Myra Mine. ARD is either being prevented or is controlled at these operating mines (Errington and Ferguson, 1987).

1.7.4 Spoil Piles

Spoil piles are wastes generated by coal strip mining in order to gain access to the coal deposit. Sulphur exists in three chemical forms in coal-bearing rocks: sulphides, sulphate and organic sulphur. Pyrite (FeS_2) is the most predominant acid producing sulphur mineral present within coal and the overburden strata. A general feature of coal overburden strata is the variability in geochemical properties. A number of different lithological units may be present with acid generation potentials varying from acid generating to highly neutralizing. This leads to extreme variation in drainage quality from spoil piles and difficulties in predicting whether spoil piles will be acid generating or not.

1.7.5 Tailings

The potential for acid generation in sulphide rich tailings has long been recognized. The low permeability of many tailings deposits and the flooding that occurs in both operating and abandoned tailings impoundments limits the rate of ARD generation and release, particularly during operation and if the tailings are deposited at high pH. Thus the full potential effect of the very large deposits of more recent acid generating tailings has not yet developed. Many of these facilities are still under active use and maintenance and any potential impacts of ARD is controlled. The concern is often that ARD may occur after abandonment. Tailings from older mines, which have been distributed along water courses as a result of direct discharge to the streams, or of impoundment failure, are responsible for substantial ARD loadings at several locations in the USA and Canada. ARD is reported to have occurred from tailings at four mines in BC (Steffen Robertson and Kirsten, 1988).

1.7.6 Stockpiles and Spent Heap-leach Piles

Low grade ore stockpiles are often of particular concern to regulatory agencies since falling metal prices may conceivably cause low grade ore to become waste. These stockpiles are often a concentrated source of ARD. Heap-leach operations include; cyanide leach for gold recovery, acid-leach for base-metal recovery and bioleach processes which are also generally low-pH. These operations generally result in piles of spent ore after operations have ceased. The spent heap-leach piles can be a source of ARD, particularly those associated with low pH leachates. Although cyanide leach operations are carried out at high pH, the neutralizing potential of the residual leachate may become depleted with time, resulting in a net acid generating potential for the waste.

1.8 OBJECTIVES OF PREDICTION, CONTROL AND MONITORING TECHNOLOGY

The economic and environmentally safe development of mines in sulphide-bearing rock is dependent on confident prediction of the acid generating potential of the ore and mine wastes. Accurate prediction of the acid generation potential of wastes is essential in order to determine the level of ARD control required. This information is necessary in order to optimize the design of control measures and the waste management operating plan.

The objective of ARD control is to achieve the necessary control to satisfy environmental criteria using the most cost effective technique. It is important to distinguish between the prevention of acid rock drainage for proposed operations and the abatement of acid rock drainage at existing facilities after the problem is manifest. In this Guide, the term 'prevention' is used to describe control designed for a proposed, potential ARD producing facility, while 'abatement' is used to describe control implemented at a facility already generating ARD. The term 'control' is used collectively to include prevention and/or abatement measures.

The technical options available for the control of contaminated drainage are considerably greater at proposed rather than at existing operations. This is because control measures are limited by the site specific and waste deposit conditions in existence at established mines. Further, if the problem was not anticipated before commencement of mining, provision for the cost of control measures are not included in the financial planning of an existing mine. In these cases the economic impacts can be far greater than if control measures were included in the design. This demonstrates the importance of acid generation prediction in the prevention of acid rock drainage.

An important criterion that needs to be identified before designing ARD control is the length of time that the measure needs to remain effective. Certain measures or techniques are effective short term controls but may not be suitable for the long-term control of ARD. Conversely, the most suitable long-term control method may not be the most cost-effective or practical short-term measure.

There are at present three generally accepted approaches to the prevention or control of acid rock drainage, as follows:

1) Control of the acid generation process.
2) Control of acid rock drainage migration.
3) Collection and treatment of acid rock drainage.

The above three control categories are listed in order of preference. It is best to first control acid generation if possible. Where acid generation is not prevented, contaminant migration control should be implemented. If neither of these control measures are in effect, it is necessary to collect and treat ARD. A combination of measures from one or more of these categories may provide the most secure ARD control.

As a consequence of the severe potential environmental impact of ARD, the uncertainty of some of the prediction and control techniques, and the possible long period of time before ARD is realized, regular and extensive monitoring is required. The objectives of ARD monitoring are to:

- Detect the onset of acid generation before ARD develops to the stage where environmental impact occurs. Abatement measures, if required, need to be implemented as soon as possible.

- Monitor the effectiveness of control measures.

CHAPTER 2: THE ACID GENERATION PROCESS

2.1 INTRODUCTION

Acid generation is caused by the exposure of rock containing certain sulphide minerals, most commonly pyrite (FeS_2), to air and water, resulting in the production of acidity and elevated concentrations of sulphate and metals. This is a consequence of the oxidation of sulphur in the mineral to a higher oxidation state and, if aqueous iron is present and unstable, the precipitation of ferric iron with hydroxide. Sulphide minerals are ubiquitous in the geological environment, but are primarily found in rock that lies beneath a mantle of soil and often beneath the water table. Under natural conditions, the overlying soil and groundwater minimize contact with oxygen and thus allow acid generation to proceed at such a low rate that the effect on background water quality is negligible or not detectable. The exposure of this rock to air and water, such as a result of road building, general surface disturbance, or mining, may accelerate the rate of acid generation and cause an impact on the environment under certain conditions (Section 2.3).

Acid rock drainage (ARD) has likely been known to mankind since the mining of sulphide-bearing rock began (Paine 1987). However, the factors that cause ARD and the impact on the environment have been studied in detail only in the last few decades. It is now known that certain conditions, including the right combination of mineralogy, water, and oxygen, are necessary for the development of acid rock drainage. Such conditions are not always met and, consequently, acid rock drainage is not found at all mines with sulphide-bearing material.

The ability of a particular rock sample to generate net acidity is a function of the relative content of acid generating minerals and acid consuming minerals. The process by which acid is consumed is known as "neutralization". Acid waters generated by sulphide oxidation in a rock may be neutralized upon contact with acid-consuming minerals. As a result, the water draining from the rock may have a neutral pH and negligible acidity despite on-going sulphide oxidation. However, if the acid consuming minerals are dissolved, washed out, or coated by other minerals through encapsulation, then, as acid generation continues, acid water may eventually drain from the rock mass. Whenever the acid consuming capability of a rock (the "neutralization potential") exceeds acid generation capability, it is generally considered that any acid will be consumed and water draining from the rock will be at neutral pH. There are limitations to this assumption as discussed in Section 2.4 and Chapter 4.0, Section 4.4.

It is important to consider the scale of examination when addressing acid drainage. A rock which produces pH-neutral conditions in water passing over it may have acid generation occurring in microenvironments around sulphide grains. The resulting acidic water may be neutralized by the remainder of the sample as it leaves the microenvirenment. If an acid-generating rock has no neutralization potential, then there will be acidic water present on the scale of both the sulphide grains as well as the rocks as a whole. A more complex situation occurs when pH-neutral water flowing over a rock invades and flushes the microenvironments of the sulphide grains, resulting in a slower rate of acid generation and a slower consumption of neutralization potential. For clarity in terminology, "acid generation" generally refers to the reaction in the microenvironment around a sulphide grain whereas "acid mine drainage" refers to the chemical composition of water on a larger scale emanating from a rock or waste-rock pile.

The time interval between initial disturbance of rock and the peak rate of acid generation may range from days to years, depending on a number of environmental factors and the neutralization potential of the rock. In addition, the rate at which acid generation occurs through time will vary depending on several environmental factors and geochemical characteristics of the sulphide minerals. As a result, acid generation is not a simple process, rather it is a complex set of chemical reactions changing though time, which are currently the topic of a large amount of scientific research.

2.2 SULPHIDE MINERALS

Crystalline substances which contain sulphur combined with a metal (for example, Fe) or semi-metal (for example, As) but no oxygen are called sulphide minerals (Table 2.2-1). If a metal or semi-metal are both present in a mineral (for example, Arsenopyrite FeAsS) the semi-metal substitutes for sulphur in the crystal structure. These minerals form in strongly anoxic (that is, chemically reducing) environments as indicated by sulphur which is present in its lowest natural oxidation state. In oxygenated environments, sulphur exists in higher oxidation states such as $S_2O_3^{2-}$, SO_3^{2-}, and SO_4^{2-} (sulphate) and forms minerals with oxygen (for example, gypsum, $CaSO_4.2H_2O$).

Under certain geological conditions, most notably near-surface, low temperature deposits (bogs and swamps, etc.) and/or rapid deposition (eg. mid-oceanic ridge sulphide deposits), sulphide may be precipitated in amorphous (non-crystalline) or poorly crystalline forms. For iron-sulphide minerals, amorphous FeS or greigite (Fe_3S_4, Table 2.2-1) may form initially and then alter to pyrite through sulphurization (Sweeney and Kaplan 1973). This process may lead to the formation of raspberry-like balls or "framboids" of fine-grained pyrite crystals. This framboidal pyrite has a significantly higher rate of acid generation when exposed to an oxidizing environment than coarse-grained, euhedral pyrite (Carrucio et al. 1977)

Marcasite (Table 2.2-1) is a low-temperature iron-sulphide mineral that may form instead of pyrite and which reportedly has a higher rate of acid generation under oxidizing conditions than crystalline pyrite (Hurlbut and Klein 1977). Marcasite may also be found in higher-temperature paleoenvironments where it is metastable with respect to pyrite at temperatures greater than 157°C.

At elevated temperatures, sulphide may be mobile, leading to re-crystallization as a massive sulphide sometimes found at metal mines in British Columbia. The rate of acid generation may be relatively slow from massive sulphide, but the rate may be

TABLE 2.2-1
SUMMARY OF COMMON SULPHIDE MINERALS AND THEIR OXIDATION PRODUCTS

Mineral	Composition	Aqueous End Products of Complete Oxidation[a]	Possible Secondary Minerals Formed at Neutral PH After Complete Oxidation and Neutralization[b]
Pyrite	FeS_2	Fe^{3+}, SO_4^{2-}, H^+	Ferric hydroxides and sulphates; gypsum.
Marcasite	FeS_2	Fe^{3+}, SO_4^{2-}, H^+	Ferric hydroxides and sulphates; gypsum.
Pyrrhotite	$Fe_{1-x}S$	Fe^{3+}, SO_4^{2-}, H^+	Ferric hydroxides and sulphates; gypsum.
Smythite, Greigite	Fe_3S_4	Fe^{3+}, SO_4^{2-}, H^+	Ferric hydroxides and sulphates; gypsum.
Mackinawite	FeS	Fe^{3+}, SO_4^{2-}, H^+	Ferric hydroxides and sulphates; gypsum.
Amorphous	FeS	Fe^{3+}, SO_4^{2-}, H^+	Ferric hydroxides and sulphates; gypsum.
Chalcopyrite	$CuFeS_2$	Cu^{2+}, Fe^{3+}, SO_4^{2-}, H^+	Ferric hydroxides and sulphates; copper hydroxides and carbonates; gypsum.
Chalcocite	Cu_2S	Cu^{2+}, SO_4^{2-}, H^+	Copper hydroxides and carbonates; gypsum.
Bornite	Cu_5FeS_4	Cu^{2+}, Fe^{3+}, SO_4^{2-}, H^+	Ferric hydroxides and sulphates; copper hydroxides and carbonates; gypsum.
Arsenopyrite	FeAsS	Fe^{3+}, AsO_4^{3-}, SO_4^{2-}, H^+	Ferric hydroxides and sulphates; ferric and calcium arsenates; gypsum.
Realgar	AsS	AsO_4^{3-}, SO_4^{2-}, H^+	Ferric and calcium arsenates; gypsum.
Orpiment	As_2S_3	AsO_4^{3-}, SO_4^{2-}, H^+	Ferric and calcium arsenates; gypsum.
Tetrahedrite and Tennenite	$Cu_{12}(Sb,As)_4S_{13}$	Cu^{2+}, SbO^{3-}, AsO_4^{3-}, SO_4^{2-}, H^+	Copper hydroxides and carbonates; calcium and ferric arsenates; antimony materials; gypsum.
Molybdenite	MoS_2	MoO_4^{2-}, SO_4^{2-}, H^+	Ferric hydroxides; sulphates; molybdates; molybdenum oxides; gypsum.
Sphalerite	ZnS	Zn^{2+}, SO_4^{2-}, H^+	Zinc hydroxides and carbonates; gypsum.
Galena	PbS	Pb^{2+}, SO_4^{2-}, H^+	Lead hydroxides, carbonates, and sulphates; gypsum.
Cinnabar	HgS	Hg^{2+}, SO_4^{2-}, H^+	Mercuric hydroxide; gypsum.
Cobaltite	CoAsS	Co^{2+}, AsO_4^{3-}, SO_4^{2-}, H^+	Cobalt hydroxides and carbonates; ferric and calcium arsenates; gypsum.
Niccolite	NiAs	Ni^{2+}, AsO_4^{3-}, SO_4^{2-}, H^+	Nickel hydroxides and carbonates; ferric, nickel and calcium arsenates; gypsum.
Pentlandite	$(Fe,Ni)_9S_8$	Fe^{3+}, Ni^{2+}, SO_4^{2-}, H^+	Ferric and nickel hydroxides; gypsum.

---

[a] Intermediate species such as ferrous iron (Fe^{2+}) and $S_2O_3^{2-}$ may be important.

[b] Depending on overall water chemistry, other minerals may form with, or instead of, the minerals listed here.

accelerated during the mining process through blasting and grinding. In general, the relative rates of oxidation for sulphide minerals (Table 2.2-1) under typical environmental conditions are undefined and detailed experimentation is recommended.

Oxidation of these minerals may lead to the formation of secondary minerals after some degree of pH neutralization or if pH is maintained near neutral during oxidation. Some of these minerals are listed in Table 2.2-1, although other minerals may form in addition to, or instead of these, minerals depending on water chemistry, extent of oxidation, and the presence of other compounds such as aluminosilicates. These secondary minerals may encapsulate the sulphide mineral and/or any neutralizing mineral, slowing the reaction rate.

2.3 CHEMICAL AND BIOLOGICAL REACTIONS RELATED TO ACID GENERATION

Acid generation, as well as acid consumption, are the result of a number of interrelated chemical reactions. The primary ingredients for acid generation are: (1) sulphide minerals, (2) water or a humid atmosphere, and (3) an oxidant, particularly oxygen from the atmosphere or from chemical sources. Exclusion of absolutely all moisture or oxidant will stop acid generation. In most cases, bacteria play a major role in accelerating the rate of acid generation and the inhibition of bacterial activity in these cases will lessen the rate of acid generation.

The reactions of acid generation are best illustrated by examining the oxidation of pyrite (FeS_2) which is one of the most common sulphide minerals. The first important reaction is the oxidation of the sulphide mineral into dissolved iron, sulphate, and hydrogen (H^+):

$$FeS_2 + 7/2\ O_2 + H_2O \text{ --> } Fe^2+ + 2SO_4^{2-} + 2H^+ \qquad (2.1)$$

(Solid + gas + water --> ferrous iron, sulphate, and H^+ in water)

The dissolved Fe^{2+}, SO_4^{2-}, and H^+ represent an increase in the total dissolved solids and acidity of the water and, unless neutralized, the increasing acidity is often associated with a decrease in pH. If the surrounding environment is sufficiently oxidizing, much of the ferrous iron will oxidize to ferric iron:

$$Fe^{2+} + 1/4\ O_2 + H^+ \text{ <--> } Fe^{3+} + 1/2\ H_2O \qquad (2.2)$$

At pH values above 2.3 to 3.5, the ferric iron will precipitate as $Fe(OH)_3$, leaving little Fe^{3+} in solution while lowering pH at the same time:

$$Fe^{3+} + 3H_2O \text{ <--> } Fe(OH)_3 \text{ (solid)} + 3H^+ \qquad (2.3)$$

Any Fe^{3+} from Reaction 2.2 that does not precipitate from solution through Reaction 3 may be used to oxidize addition pyrite:

$$FeS_2 + 14Fe^{3+} + 8H_2O \text{ --> } 15Fe^{2+} + 2SO_4^{2-} + 16H^+ \qquad (2.4)$$

Based on these simplified basic reactions acid generation that produces iron which eventually precipitates as $Fe(OH)_3$ may be represented by a combination of Reactions 2.1, 2.2, and 2.3:

$$FeS_2 + 15/4\ O_2 + 7/2\ H_2O \dashrightarrow Fe(OH)_3 + 2SO_4^{2-} + 4H^+ \qquad (2.5)$$

On the other hand, the overall reaction for stable ferric iron that is used to oxidize more pyrite (combinations of Reactions 2.1, 2.2, and 2.4) is:

$$FeS_2 + 15/8\ O_2 + 13/2\ Fe^{3+} + 17/4\ H_2O \dashrightarrow 15/2\ Fe^{2+} + 2SO_4^{2-} + 17/2\ H^+ \qquad (2.6)$$

All of the above equations, except 2.2 and 2.3, assume the oxidized mineral is pyrite and the oxidant is oxygen. However, other sulphide minerals such as pyrrhotite, (FeS) and chalcocite (Cu_2S) have other ratios of metal:sulphide and metals other than iron (Section 2.2). Other oxidants and other sulphide minerals also have different reaction pathways, stoichiometries, and rates, but research on these variations is limited.

The primary chemical factors which determine the rate of acid generation are:

a) pH;
b) Temperature;
c) Oxygen content of the gas phase, if saturation is less than 100%;
d) Oxygen concentration in the water phase;
e) Degree of saturation with water;
f) Chemical activity of Fe^{3+} (see Reaction 2.4);
g) Surface area of exposed metal sulphide; and
h) Chemical activation energy required to initiate acid generation.

Certain bacteria may accelerate or decelerate the rate at which some of the above reactions proceed (Table 2.3-1), thereby increasing or decreasing the rate of acid generation. <u>Thiobacillus ferrooxidans</u> in particular is known to accelerate Reactions 2.2 and 2.4 through its enhancement of the rate of ferrous-iron oxidation (e.g., Kleinman and Erickson 1983). <u>T. ferrooxidans</u> may also accelerate Reaction 2.1 through its enhancement of the rate of reduced-sulphur oxidation. Experimental testing of the many other bacterial species capable of oxidizing iron and sulphur is generally limited (Table 2.3-1). In addition several species are known to reduce sulphur and iron, potentially counteracting acid generation.

Most testing of <u>T. ferrooxidans</u> has involved oxidation of pyrite (FeS_2); however the bacterium may accelerate oxidation of sulphides of antimony, gallium, molybdenun, arsenic, copper, cadmium, cobalt, nickel, lead and zinc (Lundgren and Silver, 1980).

For bacteria to thrive, environmental conditions must be favourable; for example, <u>T. ferrooxidans</u> is most active in waters with pH around 3.2. If conditions are not favourable, the bacterial influence on acid generation will be minimal. This apparent importance of environmental conditions explains the contradiction in reported experimentation that show bacterial influence ranges from major to negligible. Experimental laboratory-based and in-field tests with bactericides indicate bacterial activity enhances the rate of acid generation (as indicated by sulphate and acidity) by factor of up to five with one extreme measurement of a factor of 20 (See Chapter 6.0, Section 6.7).

TABLE 2.3-1

BACTERIAL SPECIES WHICH INFLUENCE RATE OF SULPHUR AND IRON OXIDATION

Bacteria Species	Type	Optimal Growth Chemical Environment	Reference
Thiobacillus ferro-oxidans	Sulphur oxidising Iron oxidising	pH = 2.5 - 3.5	-
T. novellus	Sulphur oxidising	pH = neutral to alkaline	-
T. thioporus	Sulphur oxidising		
T. denitiricans	Sulphur oxidising	pH = neutral to alkaline Nitrate supply for reduction to N_2	-
Arthrobacter sp.	Sulphur oxidising	-	Wallis and Ladd (1984)
Bacillus sp.	Sulphur oxidising	-	
Flavobacterium sp.	Sulphur oxidising	-	
Pseudomonas sp.	Sulphur oxidising	-	
Desulfavibrio sp.	Sulphur reducing	-	Wallis and Ladd (1984)
Desulfotomaculum sp.	Sulphur reducing	-	
Salmonelis sp.	Sulphur reducing	-	
Proteus sp.	Sulphur reducing	-	
Sulfolosusp.	Iron oxidising	-	Wallis and Ladd (1984)
Metalloginium sp.	Iron oxidising	-	
Sidenocapsa sp.	Iron oxidising	-	
Leptothrax sp.	Iron oxidising	-	
Gallionella sp.	Iron oxidising	-	
Vibrio sp., Bacillus sp.	Iron reducing	-	
Aerobacter aerogenus	Iron reducing	-	

In situations where bacterial acceleration is significant, there are additional factors which determine the bacterial activity and the associated rate of acid generation:

a) Biological activation energy;
b) Population density of bacteria;
c) Rate of population growth;
d) Nitrate concentration;
e) Ammonia concentration;
f) Phosphorus concentration;
g) Carbon dioxide content; and
h) Concentrations of any bacterial inhibitors.

The bacterial contribution to sulphide and iron oxidation can be complex through contributions from many species. However, research indicates that Thiobaccillus ferrooxidans often plays a major role in organic-enhanced oxidation in natural environments. This justifies the attention payed to this bacterium (Section 2.3), but does not always justify the exclusion of other bacteria from consideration.

Following the oxidation of a sulphide mineral, the resulting acid products may either be immediately flushed away by water moving over the rock or, if there is no water movement, accumulate in the rock while remaining readily available for flushing. If the acid products are flushed away from the sulphide mineral, they may eventually encounter an acid consuming mineral; the resulting neutralization will remove a portion of the acidity and iron from solution and will neutralize pH. Sulphate concentrations will usually not be affected by neutralization unless mineral saturation with respect to gypsum is attained. Consequently, sulphate may sometimes be used as an overall indicator of the extent of acid generation even after neutralization by acid consuming minerals has occurred.

The most common acid consuming mineral is calcite ($CaCO_3$), which consume acidity through the creation of HCO_3^- or $H_2CO_3^0$:

$$CaCO_3 + H^+ \text{-->} Ca^{2+} + HCO_3^- \qquad (2.7)$$

and

$$CaCO_3 + 2H^+ \text{-->} Ca^{2+} + H_2CO_3^0 \qquad (2.8)$$

There are also other acid consuming minerals such as $Al(OH_3)$ (Table 2.3-2):

$$Al(OH)_3 + 3H^+ \text{-->} Al^{3+} + 3H_2O \qquad (2.9)$$

It is not unusual for a rock to contain both sulphide minerals and acid consuming minerals. The balance between the two types will determine whether the rock will eventually produce acid conditions in the water passing over and through the rock, and this balance forms the basis of the experimental procedure used in static tests (Chapter 4.0, Section 4.4).

TABLE 2.3-2

SUMMARY OF ACID CONSUMING MINERALS AND THEIR NEUTRALIZING CHARACTERISTICS

MINERAL	COMPOSITION	ACID CONSUMING[a] POTENTIAL	BUFFER pH
Calcite, Aragonite	$CaCO_3$	100	5.5 - 6.9
Siderite	$FeCO_3$	116	5.1 - 6.0
Magnesite	$MgCO_3$	84	-
Rhodochrosite	$MnCo_3$	115	-
Witherite	$BaCO_3$	196	-
Ankerite	$CaFe(CO_3)_2$	108	-
Dolomite	$MgCa(CO_3)_2$	92	-
Malachite	$Cu_2CO_3(OH)_2$	74	5.1 - 6.0
Gibbsite	$Al(OH)_3$	26	4.3 - 3.7
Limonite/Goethite	FeOOH	89	3.0 - 3.7
Manganite	MnOOH	88	-
Brucite	$Mg(OH)_2$	29	-

[a] Acid consuming potential is given as the weight (g) of the mineral required to have the same neutralising effect as 100 g of calcite. For example, the mole weight of siderite is 116 g and the mole weight of calcite is 100 g. Therefore, 116 g of magnesite are required to supply the same amount of alkalinity as 100 g of calcite although the two minerals will not necessarily neutralize low pH waters to the same pH (see buffer pH column).

2.4 STAGES IN THE DEVELOPMENT OF ACID ROCK DRAINAGE

The development of acid drainage is a complex combination of inorganic and sometimes organic processes and reactions. In order to produce severe acid drainage (pH <3), sulphide minerals must create an optimum microenvironment for rapid oxidation and must continue to oxidize long enough to exhaust the neutralization potential of the rock.

As a rock with reactive sulphide minerals is initially exposed to flowing water and oxygen, sulphide oxidation and acid generation begins. Any calcium-based carbonate such as calcite ($CaCO_3$) in the rock neutralizes this small amount of acidity and maintains neutral to alkaline (pH >7) in the water passing over the rock. As acid generation continues and the calcium-based carbonate is consumed or becomes unavailable, the pH of the water decreases, which in turn enhances the conditions for acid generation (Figure 2.4-1). As the rate of acid generation accelerates, the pH progressively decreases in a theoretical step-like manner. Each plateau of relatively steady pH represents the dissolution of a neutralizing mineral that becomes soluble at that pH. If the rate of acid generation remains sufficiently strong until all neutralization potential is removed, then pH values below 3 will occur and will reflect the combined influence of all the factors listed in Section 2.3. At some time in the future, decades and possibly centuries from the initiation of acid generation, the rate will slow as the more reactive sulphide has completely oxidized and pH will increase until the rock becomes essentially inert and the ambient pH of the water is not affected. The time scale for each successive stage may be days to years to centuries and is dependant on the factors required for acid generation.

The step-like degradation of pH through dissolution of neutralizing minerals has been defined through field studies and computer simulations (Morin et al. 1988a and b; Morin and Cherry, 1988). Based on evaluation of acidic seepage from six tailings impoundments in Canada and the U.S.A. and from one series of laboratory-based column experiments (Morin, 1988) the pH values (plateaus) expected from the dissolution of various neutralizing minerals are:

Mineral	pH Range
Calcium-based carbonate	5.5-6.9
Siderite (iron-based carbonate)	5.1-6.0
Aluminum hydroxide	4.3-5.0
Iron hydroxide	3.0-3.7

For example, if acid generation persists at a sufficient rate to remove all neutralizing minerals except iron hydroxide, the pH of the water will be around 3.0 to 3.7 and will then return to the ambient water pH as the rate becomes negligible. This type of behaviour has been noted in numerous laboratory-based experiments for a proposed gold mine in British Columbia (City Resources, 1988) and for coal mines in Maryland and Pennsylvania (Helz et al, 1987).

The step-like degradation of pH highlights the care that must be taken in any type of "batch" test where a pre-defined amount of rock is mixed with a pre-defined quantity of acid. The resulting neutralization by the rock will simply reflect the plateau that is reached with that particular ratio of solid:liquid (Morin and Cherry, 1988). A greater proportion of acid may lower the pH to the next plateau or completely remove all neutralizing minerals. For this reason, it is important to define the initiation, variation in rate, and duration of acid generation using kinetic tests (Chapter 4.0, Section 4.5).

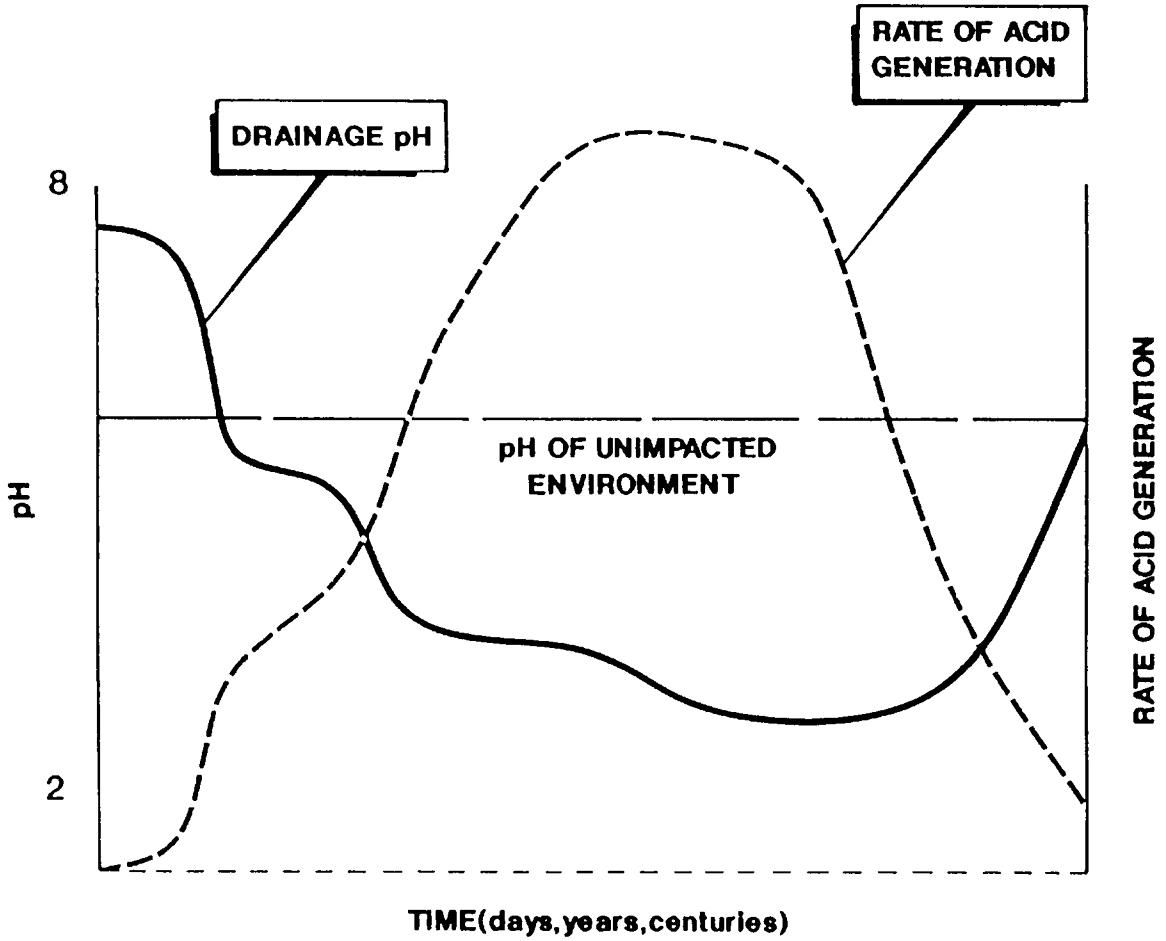


FIGURE 2.4-1
SCHEMATIC DIAGRAM OF ACID-DRAINAGE PRODUCTION

CHAPTER 3: METAL LEACHING AND MIGRATION PROCESSES

3.1 INTRODUCTION

The low pH pore water that occurs as a consequence of acid generation is capable of mobilizing heavy metals and other soluble constituents contained within the waste. The major environmental impact of ARD is not realized until this poor quality water migrates away from the site of generation and enters the receiving environment. It is the high metal loadings in the water emerging from the waste which is most harmful to the environment.

A number of reactions occur along the route as the water, initially of low pH, migrates from the source to the receiving environment. The quality of pore water which develops as a consequence of acid generation is dependent on the nature of the sulphides, the availability and type of soluble constituents in the waste, the nature of alkaline reactants, and the physical properties of the waste. Paine (1987) presents some examples of ARD quality obtained from a variety of Canadian tailings, waste dumps and underground mines as shown in Table 3.1-1.

TABLE 3.1-1

EXAMPLES OF ACID ROCK DRAINAGE QUALITY
(After Paine, 1987)

Parameter[1]	Seepage from Abandoned Uranium Mine Tailings Pond in Ontario	Waste Rock Dump Seepage From Active Silver Mine in British Columbia	Minewater from Underground Copper Mine in British Columbia
pH	2.0	2.8	3.5
Sulfate	7,440	7,650	1,500
Acidity	14,600	43,000	--
Iron	3,200	1,190	10.6
Manganese	5.6	78.3	6.4
Copper	3.6	89.8	16.5
Aluminum	588	359	--
Lead	0.67	2	0.1
Cadmium	0.05	0.5	0.143
Zinc	11.4	53.2	28.5
Arsenic	0.74	25	0.05
Nickel	3.2	8.0	0.06

[1] units are mg/l except pH

There are a number of naturally occurring physical, chemical and biological properties of mine waste facilities that control metal solubility and contaminant migration. The mobilization of metals is principally controlled by chemical factors while the processes that occur along the migration route are controlled by physical and chemical factors. This chapter describes these controls and how they influence ARD quality and the time dependent evolution of ARD. The reactions that occur when ARD reaches the receiving environment and how this impacts the environment are discussed to complete the chapter.

3.2 PHYSICAL CONTROLS

There are certain physical characteristics that influence metal solubility such as the waste particle size and shape, temperature, and pressure of pore gases. However, these properties are generally secondary to chemical factors in the reactions that occur during mobilization of metals. The influence of physical properties are more important in the rate of ARD migration and in the reactions that occur along the route of migration. Important characteristics in this regard include climatic conditions, waste permeability, availability of pore water, pore water pressure, and the processes or mechanisms of movement, whether by stream flow or diffusion. These factors control the rate of movement of contaminant fronts, the amount of dilution and the degree of mixing that occurs as the contaminants move from the source to the receiving environment. The physical properties in the waste material are likely to be different from those in the underlying subsurface strata. A number of contaminant fronts may therefore occur, all moving at different rates. In hard rock waste dumps, for example, surface water yields tend to occur before ground water yields due to lower retardation and hence rapid migration through the waste rock. The water quality of the yields, either surface or ground water, are a function of the dilution and buffering reactions that occur en-route. It is extremely important to consider these factors in order to understand the likely evolution of ARD from any particular waste.

3.3 CHEMICAL CONTROLS

The solubility of metals is generally controlled by the pH of the leachate, however, other chemical controls include Eh, adsorption phenomena, the type of metal being mobilized and the chemical composition of the leachate. The water quality of the drainage entering the receiving environment is a function of these chemical controls as they occur along the route of migration. As acid drainage moves away from the sulphide source, through the waste material, more acid generating material may be encountered causing a further reduction in pH. Conversely, the drainage may pass over alkaline material in which case partial or complete neutralization may take place. The acid generating and neutralizing chemical reactions are presented in Chapter 2.0.

With decreasing pH the dissolved metal load generally increases, however, a combination of chemical conditions could occur where metals are mobilized at neutral or even alkaline conditions. During neutralization of low pH drainage, precipitation of many of the soluble metals may occur and the resultant drainage will contain the residual metals and products of the buffering reactions.

A characteristic that has been observed for copper and massive sulphide ores in Western Canada is elevated zinc loadings in neutral drainage (Steffen Robertson and Kirsten, 1986). While dissolved copper precipitates out to low levels as ARD is neutralized, the zinc remains at relatively high concentration until the pH is raised to values above 9.5. The chemical composition of the contaminated front is therefore likely to change as it undergoes various dissolution and buffering reactions as the front moves through the waste or subsurface strata.

3.4 BIOLOGICAL CONTROLS

Biological activity along the route of migration may influence the quality of ARD. Metal leaching occurs where iron oxidizing bacteria are present with iron and metal sulphides. Biological species can also attenuate the mobility of metals by absorption or possibly precipitation. Biological activity may be a factor if the drainage passes through wetland conditions or areas of biological activity before entering the receiving environment.

3.5 IMPACT ON THE ENVIRONMENT

Acid rock drainage leaving the waste facility passes through or over soils or rock which may have a neutralization capacity. ARD also mixes with surface and ground water which neutralize and dilute the ARD. The environment therefore has a certain capacity to naturally treat the drainege which can result in some improvement of the ARD. As the store of any basic materials close to the deposit is consumed the ARD plume migrates further from the deposit, resulting in an ever increasing zone of impact. At any time there is a point some distance from the waste facility at which the effect of ARD is abated by the neutralization and dilution capabilities of the environment. In some areas, natural steams can be acidic due to weathering and oxidation of natural sulphide outcrops. In this case the impact of ARD would be different to that on neutral or basic streams. The environmental impact of ARD is therefore a function of time and is dependent on drainage quality, as well as baseline environmental conditions and the natural dilution and neutralizing capacities of the environment.

Some environments have an extremely high load of basic materials in the soils, surface and ground waters. This is the case in many of the old deeply weathered landscapes with well developed old soils. In Canada, were recent glaciation has stripped off all old soils, only relative fresh rocks and new, unweathered soils are exposed. These conditions generally yield soils, ground and surface waters which are low in neutralizing capacity. Thus the potential for extensive impact on the surface and ground waters downstream from the deposit is generally large in regions such as British Columbia. The dilution capacity of the environment depends on the magnitude of flow of the receiving waters in relation to the contaminated drainage. Stream flows are generally a function of the precipitation of the area.

For all waste deposits there is a level of ARD release that can be sustained by the environment without significant damage. It is the objective of ARD abatement measures to reduce ARD releases to below this level.

3.6 APPLICATION OF THEORY TO PREDICTION, CONTROL AND MONITORING

A description of the processes associated with acid generation, subsequent metal leaching, and finally migration of the products from the source to the environment, are described in Chapters 2.0 and 3.0 of this guide. The purpose of these chapters is to give the reader a general understanding of the processes involved during the different stages in the development of acid rock drainage. This understanding of the different processes and the factors that control them is extremely important, if not essential, in developing effective prediction, control and monitoring methods.

The success of ARD prediction methods depends on the ability to determine the potential for sulphide bearing rock to generate acid coupled with the potential of the material to neutralize acid, and the ability to determine the relative rates of these reactions. The rate of release of acidity from sulphide oxidation, and alkalinity from carbonate dissolution, may be quite different and could affect the generation of ARD. An understanding of the biochemical reactions and site specific conditions is important in developing an appropriate prediction test program.

The control of ARD may be approached in different ways. Control measures may be directed at preventing the acid generation reactions from occurring or may be aimed at preventing the transport of contaminants to the environment. An alternative approach is to allow acid generation and migration to occur and then to collect and treat the ARD before releasing it to the environment. It is important to be able to identify the source of acid drainage and the mechanisms of transport to the environment and to predict water quality in order to develop the most suitable approach to ARD control.

An understanding of the processes of ARD development is important in designing an appropriate monitoring program. Considerations in monitoring program include; chemical elements to be measured, frequency, location and method of sampling. The following chapters of this guide describe application of the theory to ARD prediction, control and monitoring.

CHAPTER 4: PREDICTION OF ACID GENERATION AND ACIDIC DRAINAGE

4.1 INTRODUCTION AND APPROACH TO PREDICTION

4.1.1 Introduction

As part of the regulatory approval process for mines in British Columbia, the government requires testing to identify potentially acid generating materials. Prior to final mine plan approval, the government will require assurance that all sources of potential acid generation have been identified and that prevention or control measures have been incorporated in mine plans where appropriate. It is important to the mining company as well as to government to do sufficient tests to ensure that problems have been addressed since remedial measures and treatment of acid drainage can be costly, with long term liabilities. The key to a successful approach to ARD testing is to do sufficient testwork to satisfy the government and the mine planner/owner to a high degree of confidence that the potential for acid generation can be predicted and controlled.

The overall procedure for evaluating the potential for acid generation at an exploration or mine site required for each stage of the Mine Development Review Process is outlined in Figure 4.1-1. This figure depicts the importance of a reliable sampling program in the overall process. Parts of this figure may apply to exploration or extraction of a bulk sample. The level of testing increases for each stage of the process.

4.1.2 Approach to Prediction

An initial step in mineral development is in the exploration stage when the property initially being investigated. The next stage is during Prospectus and Stage I phases is the development of a mine plan, defining the location, size, and management of each mine component. From the perspective of acid generation, the primary environmental components of a mine are: the mine itself, waste-rock/spoils/overburden dumps, ore stockpiles, the millsite, and tailings impoundment. These components have differing potential impacts as a consequence of differences in physical conditions, geochemical characteristics, and exposure to climatic conditions. As a result, the environmental impact of a proposed minesite must be determined on the basis of the potential impact of each component evaluated in terms of anticipated site-specific conditions.

This chapter addresses the targeting of mine components for the evaluation of their potential for acid generation and net acidity. This evaluation is accomplished through (1) comparisons with similar or nearby mines whenever possible, (2) formation of a competent program to collect representative samples at the site, (3) static tests on the samples, (4) kinetic tests based on anticipated on-site conditions using samples identified as potentially acid generating, and (5) modelling. This process is summarized in Figure 4.1-1 and is discussed in detail below.

During the exploration stage, ore deposit geology is mapped and potential ore zones are identified. The project geologist should supervise the identification and collection of representative samples of the ore and potential waste rock for acid-base accounting analyses. These results early in the mine planning stages would indicate whether acid drainage may be a potential concern at the site.

At the Prospectus Level, a preliminary mine plan is defined. The project geologist should supervise the identification and collection of representative samples of ore and waste rock for subsequent acid-base accounting analyses. The results are reported in the Prospectus to indicate whether acid drainage may be a potential concern at the site.

For Stage I studies, the project geologist should supervise the identification of homogeneous geologic units of ore and waste rock based on lithology, mineralogy, and continuity of units. If sufficient information is available from nearby mines or similar paleoenvironmental and geological areas, comparisons should be made as an initial indicator of potential acid generation (Section 4.2). A sampling program should then be implemented to collect representative samples for each geologic unit (Section 4.3). These samples are submitted for static tests (Section 4.4) to determine the potential for net acid production and to ensure that each unit is homogeneous. If the potential for net acid production is uncertailn based on the static test results, then kinetic tiests could be of value in determining the acid production potential. If the potential for net acidity is identified for any unit, the mine plan should be revised to address this issue and kinetic tests (Section 4.5) should be initiated to define the initiation of and temporal variations in the rate of acid generation and to test control/treatment options. Mathematical models may then be used to extrapolate data to longer periods of time beyond the length of the kinetic tests (Section 4.6 and Chapter 5). Predictive testwork will then aid the design, testing, and implementation of prevention/control/treatment techniques during mine operation (Chapters 6-9) and the creation of a monitoring program to observe the success of the techniques (Chapter 10).

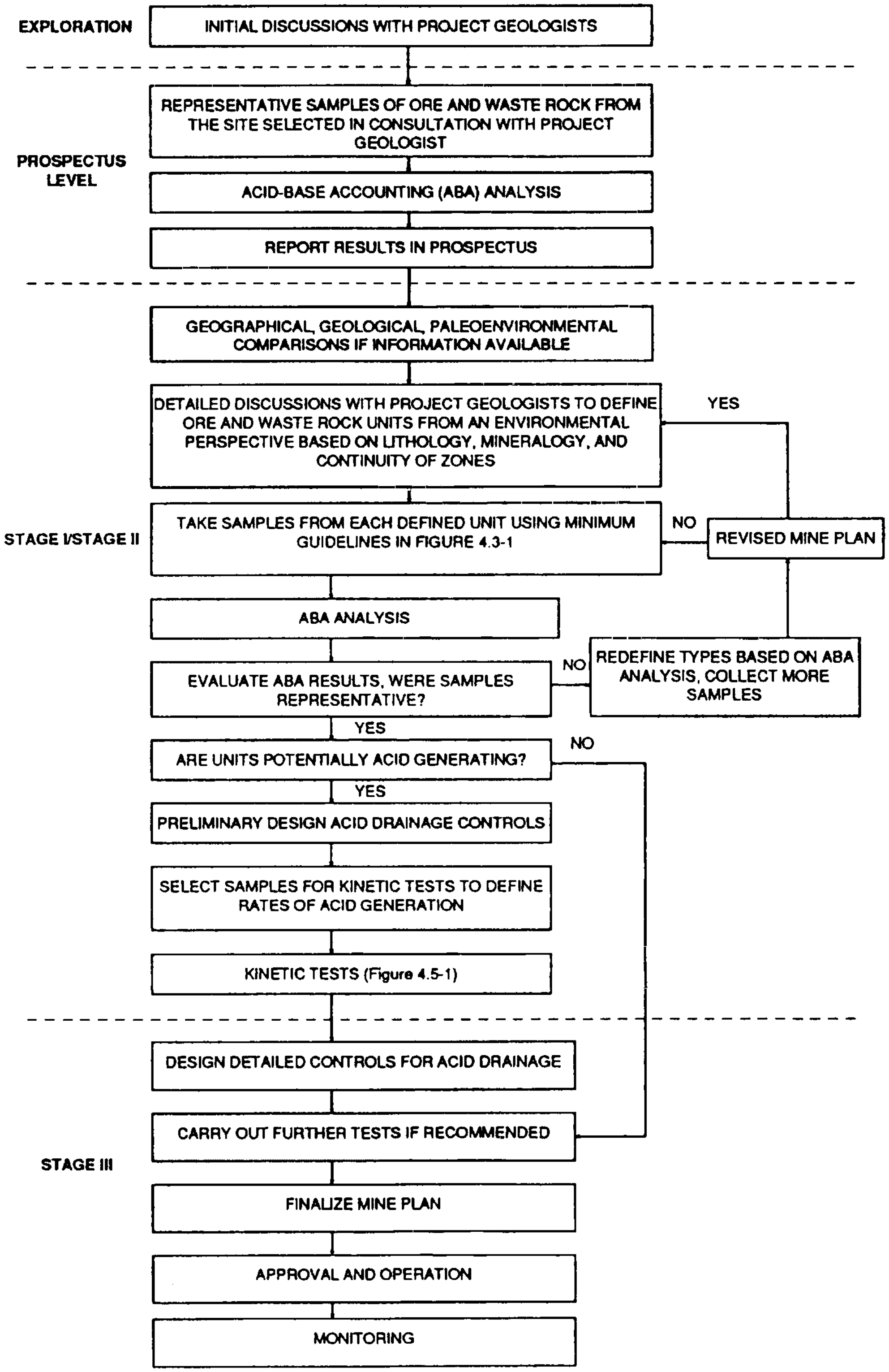


FIGURE 4.1-1
PROCEDURE FOR EVALUATING THE POTENTIAL FOR ACID GENERATION IN BRITISH COLUMBIA

4.2 GEOGRAPHICAL, PALEOENVIRONMENTAL, AND GEOLOGICAL COMPARISONS

4.2.1 Geographical Comparisons

A potential technique for the assessment of acid-generation potential involves geographical comparisons with nearby mine sites. This method is very limited in its application because it assumes that all factors which affect acid generation are constant for the mines being compared. In the case of coal mines where the same seam is being extracted, comparisons may be applicable if the footwall and hangingwall stratigraphy is identical. However, vein deposits in the same district will rarely, if ever, be comparable because the host rocks, alterations and mineralogies will be dissimilar.

At the larger scale, geographical comparisons are unlikely to be reliable because other non-geological factors (for example, physiography and climate) which affect acid generation will vary between regions.

4.2.2 Paleoenvironmental and geologic models

A basic technique for the assessment of acid-generation potential employs comparisons with other mineral deposits and mine sites which have similar paleoenvironmental or geologic characteristics. The rationale behind the method assumes that similar paleoenvironments and geologic characteristics have similar potentials for acid generation. This assumption is valid unless one or more of the numerous other factors involved in determining the potential for acid generation (Chapter 2) cause significant variations at a site.

If a mineral deposit is to be compared with other deposits for the purpose of prediction of acid rock drainage it is necessary to classify both deposits in the context of acid generation. Mineral deposit classification models exist but are typically based on economic characteristics, namely the presence or absence of mineral and metal commodities (for example, skarn deposits may be classified as precious-metal or base-metal bearing) or genetic charecteristics which control the type of commodities present (eg. high temperature or low temperature vein deposits). Some of these classifications may be useful in prediction of acid rock drainage. Errington and Ferguson (1987) showed that a simple mineral deposit classification scheme could be related to acid rock drainage in British Columbia. Carrucio et al. (1977) demonstrated that acid rock drainage could be linked to the paleoenvironment of coal formation. The latter study indicates the potential applicability of a paleoenvironmental classification of deposits to acid rock drainage prediction.

4.2.2.1 Preliminary geological classification of deposits in terms of acid rock drainage potential

A number of geological factors control whether acid rock drainage will result from mining a particular deposit. These factors, which include:

(1) oxidation state of the minerals,
(2) sulphide mineral compositions,
(3) texture and crystal development in sulphides,
(4) presence of acid-consuming minerals and
(5) presence of rock structures which increase permeability,

are partly due to the geological environment of formation. Factors (1), (2) and (4) are controlled by the chemistry of the fluid (i.e., solution or melt) from which deposition occurred, its physical characteristics and the environment of deposition. In particular, the activities of metals, sulphur, oxygen, carbon dioxide and water, Eh and pH and pressure and temperature will determine the mineral assemblage produced. The chemical variables are primarily a function of the geochemical environment in the fluid's source rocks. Although factor (3) may be a function of temperature, pressure and rate of crystallization, it is also affected by conditions after formation. Metamorphism may result in re-crystallization and mobilization of sulphides and other minerals. Finally, factor (5) is determined by structural deformation. A first step to classifying mineral deposits in terms of their potential to produce acid rock drainage should therefore be possible using these factors.

Classifications of this type must be placed in a geographical context since acid generation is due to ambient climatic conditions as well as rock characteristics. It is therefore unlikely that identical porphyry deposits in very wet and very dry regions will have the same acid generation rates.

4.2.2.2 Examples

The effect of activity of sulphur in a melt is important in the different types of magmatic deposits. In the case where sulphur is not concentrated, oxide phases such as magnetite will predominate producing a deposit with a low acid-generating potential. Higher activities of sulphur results in segregation of a sulphide melt which may eventually crystallize as pyrrhotite ($Fe_{1-x}S$) or pyrite (FeS_2). Thus, as activity of sulphur in the melt increases, later potential for acid generation also increases.

The effect of metamorphism may lead to re-classification of a deposit in terms of its acid generation potential even though geologically it still the same type. This is important in syngenetic sulphide deposits which initially consist of very fine grained poorly crystalline sulphide material which is very susceptible to oxidation and eventually, under the right conditions may recrystallize to massive pyrrhotite which is considerably less likely to generate acid.

4.2.2.3 Summary

A database to refine a geological classification scheme in terms of acid rock drainage potential should be established by government and industry. The construction of models from information in the BC AMD Task Force questionnaire (Steffen, Robertson and Kirsten, 1988) is the initial step in constructing this database.

4.3 SAMPLING METHODS FOR STATIC AND KINETIC TESTS

4.3.1 Objectives of sampling

The primary objective of a reliable sampling program is to obtain high quality representative samples of all geologic units related to the mine development plan. The units must be defined on the basis of physical and chemical homogeneity.

Although the collection of representative samples from geologic units is simple in concept, the implementation of the sampling program is more complicated and may be iterative in nature. In other words, static tests (Section 4.4) on samples from a defined geologic unit may indicate significant variability in acid-generation potential. This variability may be spatially correlated and may indicate that the unit actually consists of two or more units from the perspective of acid drainage. The additional units should then be defined, the sampling plan revised, and further samples collected to confirm the homogeneity of the new units. Such an iterative sample program may be preferable to the definition of a unit as acid-generating only in certain locations and to the implementation of special handling and control/treatment techniques for an entire unit which is not wholly acid generating. The proposed program also eliminates the need to calculate weighted averages from drill cores for the assessment of acid generation potential because a geologic unit will be defined as homogeneous.

The design of the sampling program should be directly related to the mine plan. The mine plan is developed from the exploration drilling assay data and outlines the location of the area to be mined. In an open pit the size and location of the benches is defined along with the location on the bench of the ore and waste rock, and in some cases low grade ore. In an underground mine the development workings are defined with the location of ore and waste rock.

In addition to designing the sampling program so that the samples represent the different lithological units from the perspective of acid drainage, the samples should be taken in a manner which represents the different areas in the mine plan. This approach ensures that all samples are taken from areas that will be mined or areas of wall that will be exposed in the final excavated workings. It also has the advantage that the results of acid generation prediction testwork can be directly related to timing and final placement of rock. Thus, if the test results indicate that some areas will have acid generation problems, the destination of the rock will be known.

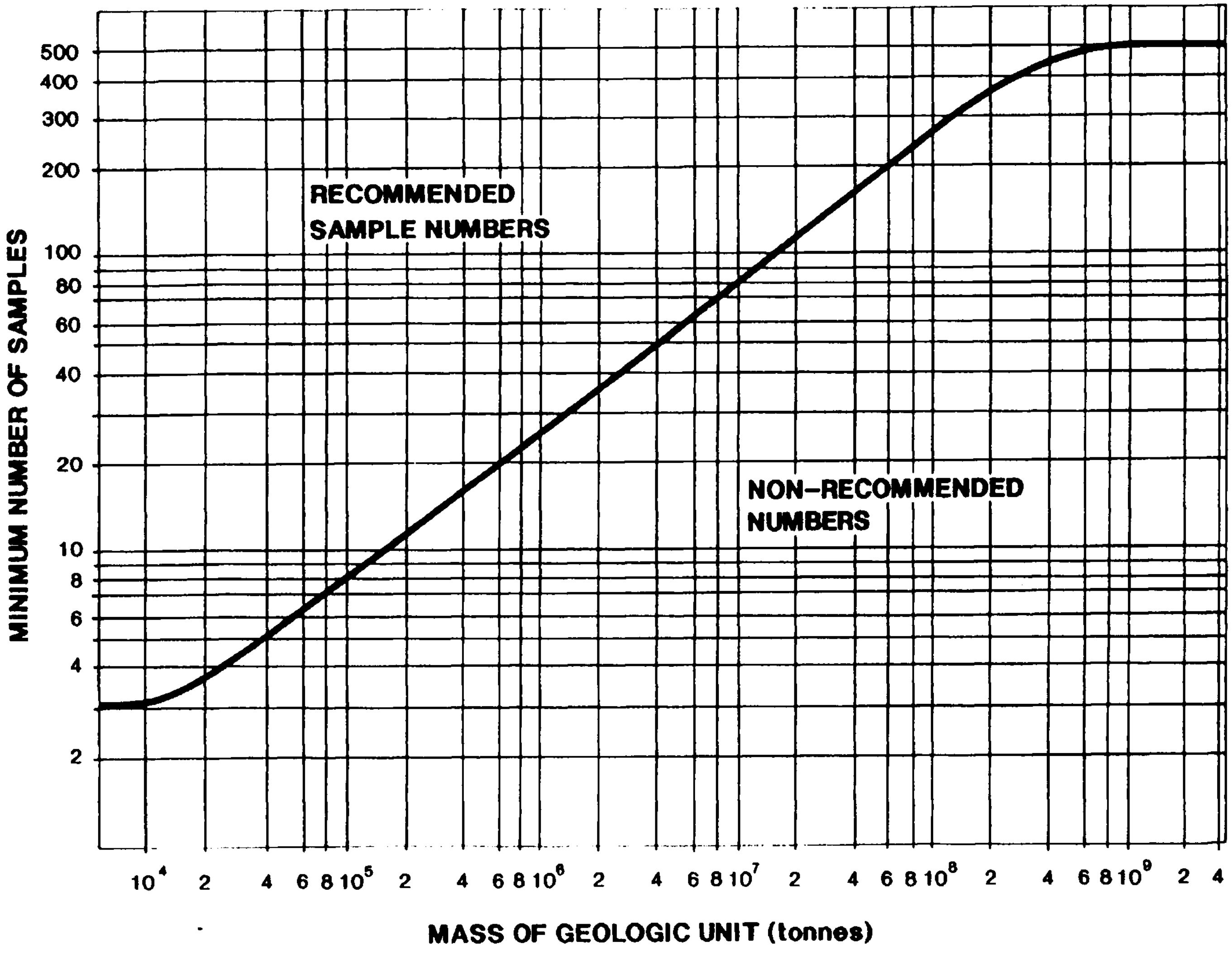


FIGURE 4.3-1
RECOMMENDED MINIMUM NUMBER OF
SAMPLES AS A FUNCTION OF MASS OF EACH GEOLOGIC UNIT
(Based on unpublished field data obtained
by Norecol Environmental Consultants)

4.3.2 Application of sampling program

The design of a sampling program begins with an understanding of the property geology at the prospectus level. The project geologist guides the initial identification of geologic units and the selection of representative samples. At the Stage I level a more detailed sampling program is required (Figure 4.3-1) if not performed for the Prospectus, in order to refine the geologic units based on homogeneity and to define more reliably the potential for net acid production. Additional information from geographical comparisons and paleoenvironment (Section 4.2) may be used in assessing the potential for net acid production. Upon any changes to the mine plan, such as the addition of a new ore zone or a change in the metallurgical process, the sampling program will expand in response.

A curve for determining the minimum number of samples to be collected to characterise each geological unit in terms of its potential to generate net acidity is being developed based on experience with a limited number of mining projects in British Columbia (Figure 4.3-1). However, the empirical and preliminary nature of the curve dictates that it should only be used as a guideline, especially when a unit is highly homogeneous. If the samples analyzed indicate a high degree of variability, the unit should be divided into two or more units and re-sampled using the minimum guideline of Figure 4.3-1. However, care must be taken when sampling a homogeneous unit with, for example, an even distribution of carbonate or sulphide veinlets. Sample size must account for any small-scale regular heterogeneity.

TABLE 4.3-2
POTENTIAL SOURCES OF SAMPLES FOR ACID GENERATION PREDICTION

MINE COMPONENT	EXISTING MINES	PROPOSED MINES
Pit walls	Drill core Pit walls	Drill core Underground exploration passages Trenches
Underground workings	Drill core Walls Excavated rock	Drill core Underground exploration passages
Waste rock/overburden piles	Waste rock piles Drill core	Drill core Underground exploration passages
Tailings	Tailings Impoundments	Pilot plant for mill process
Ore stockpiles	Ore stockpiles	Drill core Underground exploration passages
Spent ore	Heap leach	Pilot plant for heap leach

The potential sources for samples for acid generation prediction testwork are outlined in Table 4.3-2. The main source of samples for predictive testwork at proposed mine sites is drill core from exploratory drilling. Pilot plant samples should be retained to test tailings and/or spent ore. Pilot plant tailings should be sampled at regular intervals while the process is operating at equilibrium. At least three samples should be taken for characterization of tailings composition. For existing mines, samples may be obtained on-site from each component, using one or more of the methods presented in "Field Sampling Manual for Reactive Sulphide Tailings" (Canect Environmental Control Technologies, 1989).

As well as obtaining a representative sample to characterize the geologic unit, one must ensure that the sample obtained is of good integrity. The type of drill, length of time after drilling and sample storage can all affect the physical and geochemical characteristics of the samples taken from drill cores.

Ideally, samples should be diamond drill core stored in a cool, dry environment, thereby preventing weathering before testing. Also, the amount of sample required for most static and kinetic tests is relatively low (approximately 500 g), but, depending on small-scale heterogeneities such as evenly distributed veinlets, a large sample size may be desirable.

For large-scale kinetic tests such as on-site rock piles, large amounts of a specific unit are required. For proposed mines, such large volumes of specific units may not be available.

4.4 STATIC PREDICTIVE TESTS

4.4.1 Introduction

The objective of a sampling program (Section 4.3) is to provide representative samples of each geologic unit at a proposed minesite. These samples are to be analyzed in order to determine their potential for acid generation and net acidity. The first step in the analysis consists of a static test on each sample, which is rapid and relatively inexpensive.

A static test defines the balance between potentially acid-generating minerals (potential acidity) and acid-neutralizing minerals (neutralization potential) in a sample. In particular, acid-generating compounds include reactive sulphide minerals (Table 2.2-1) and acid-neutralizing compounds include carbonate minerals (Table 2.3-2). A sample will theoretically generate net acidity at some point in time only if the potential acidity exceeds the neutralization potential; otherwise the sample will not produce net acidity as long as the neutralization potential is not dissolved faster than the generation of acidity.

Despite the theoretical simplicity, static tests can not be used to predict the quality of drainage emanating from waste materials at any future time. Acid generation processes and therefore drainage quality are time-dependant and functions of a large number of complex factors such as mineralogy, rock structure and climate. For this reason, static tests shpuld be treated as a qualitative predictive method, that is they can only indicate whether or not there is a potential for generation of net acidity at some unkown time.

There are several types of static tests such as acid-base accounting, APP/sulphur ratio, and the B.C. Research Initial test. However, all of these tests are simply variations on a basic procedure and all require variations of the same basic analyses (Section 4.4.2) for determining the balance between potential acidity and neutralization potential. Consequently, the basic, common procedure will be presented and the names of the variations will be de-emphasized.

4.4.2 Procedure for conducting a static test

The initial step in defining the acid-generating/acid-neutralizing balance in a sample begins with a measurement of total sulphur in a sample (Figure 4.4-1), commonly performed with a Leco furnace/analyzer (Appendix 4.4, Section 4.4.1). The measurement of total sulphur allows the calculation of "maximum potential acidity", which may overestimate the potential for acid generation if all sulphur in a sample is not acid generating. Therefore, additional analyses may be performed to refine the potential acidity. The analyses, which are not yet proven as reliable as total sulphur, are:

1. sulphur species (Appendix 4.4, Section 4.4.2) which define short-term leachable sulphate and leachable sulphide using acid extractions; and

2. reactive sulphur (Appendix 4.4, Seotion 4.4.3) which define short-term oxidizable sulphide using hydrogen peroxide.

The unproven nature of these additional analyses places them as options in a static test (Figure 4.4-2).

Following the delineation of potential acidity, the next parameter, neutralization potential, is defined (Figure 4.4-1). The measurement of neutralization potential (Appendix 4.4, Section 4.4.5) provides a gross value for neutralization; however, this value may overestimate the capacity of the sample to neutralize the pH to an environmentally acceptable level above 6. An analysis of carbonate content (Appendix 4.4, Sections 4.4.6 and 4.4.7) will provide a more meaningful measure of neutralization potential from the perspective of pH neutralization. The carbonate analysis is a recommended as an optional part of static tests. (Figure 4.4-2)

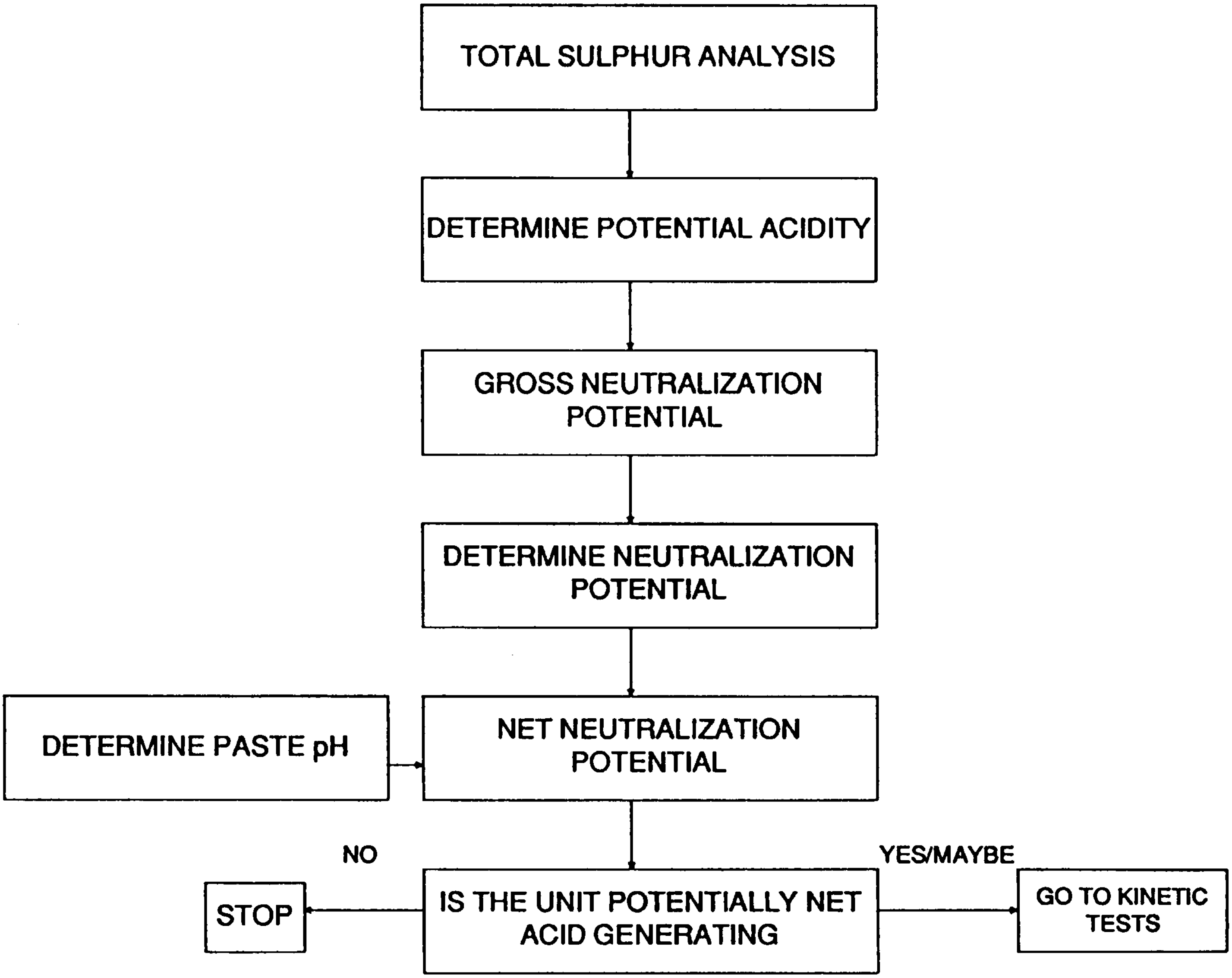
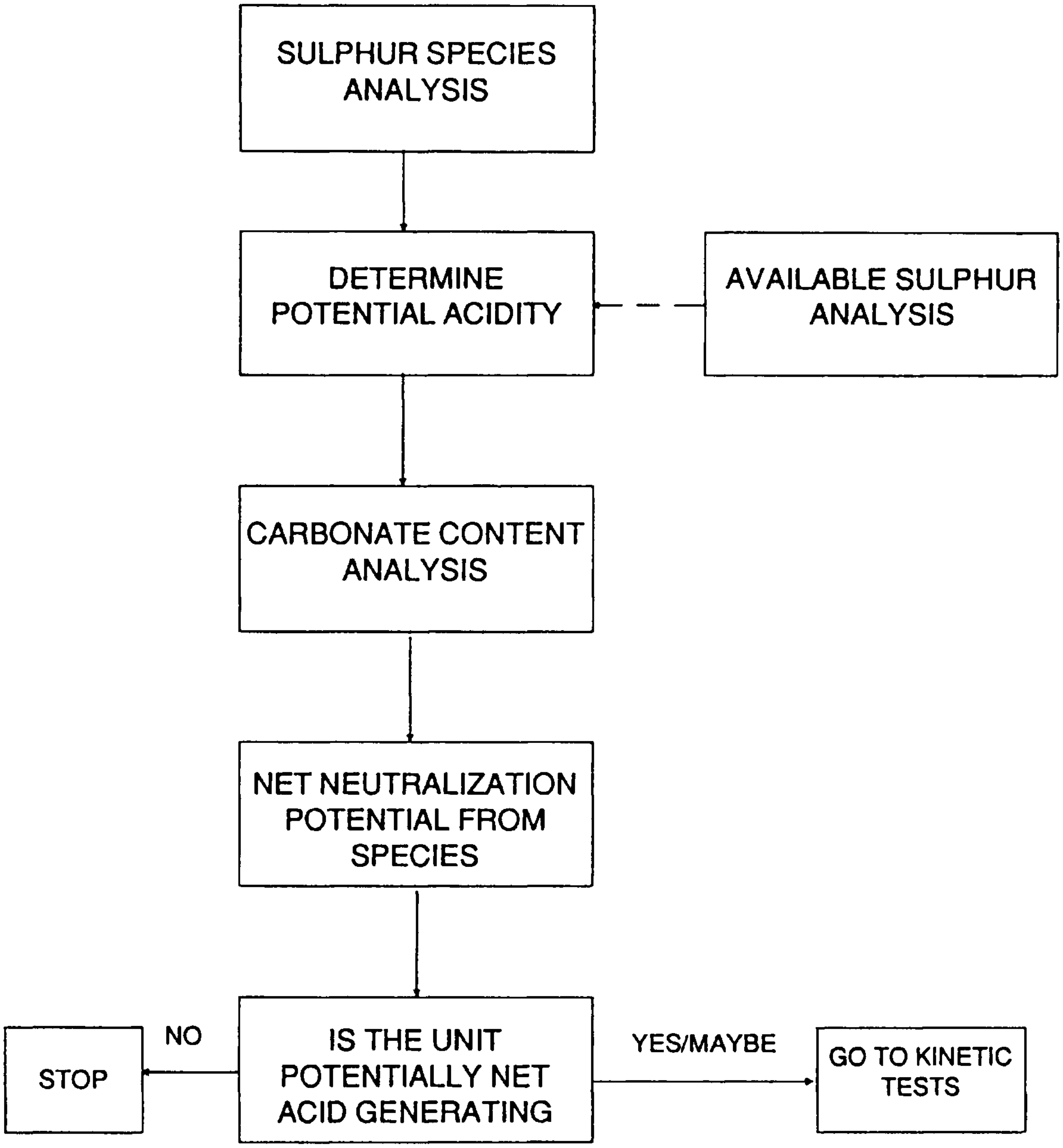


FIGURE 4.4-1
RECOMMENDED STATIC TEST PROCEDURE FOR EACH GEOLOGIC UNIT



- - -Method development required

FIGURE 4.4-2
OPTIONAL STATIC TEST PROCEDURE FOR EACH GEOLOGIC UNIT

Paste pH is measured in a paste, formed with water and the ground sample. (Figure 4.4-1). The pH value will indicate the immediate reactivity of neutralizing minerals in the sample and will indicate whether significant acid generation occurred prior to the measurement (Appendix 4.4, Section 4.4.8).

Following these analyses, the potential for net acidity is calculated by subtracting the potential acidity from the neutralization potential (Figure 4.4-1, 4.4-2) with a negative value indicating the potential for net acidity. Alternatively, a ratio of neutralization potential to potential acidity can be used (APP/sulphur ratio), but the subtraction method (acid-base accounting) is adopted here.

4.4.2.2 Interpetation of static test results

The subtraction of maximum potential acidity (based on total sulphur) from the gross neutralization potential (Figure 4.4-1) yields the "net neutralization potential (NNP)". Theoretically, a sample can be expected to generate net acidity at some point in time if the NNP is less than zero. However, based on general experience, values of NNP in the range -20 and +20 tonnes of $CaCO_3$/1000 t of sample (-2 to +2% $CaCO_3$) may be considered to have the ability to generate net acidity. This range of uncertainty is attributed to the sources of error in:

1. obtaining the objective of defining true potential acidity and neutralization;
2. converting total sulphur to acidity using a restricted conversion factor; and
3. analytical error

The subtraction of potential acidity (based on reactive sulphide) from carbonate content (Figure 4.4-2) yields the "net neutralization potential from species" (NNP(S)). This value will presumably reflect the actual net neutralization potential due to the narrower range of uncertainty and, thus, provide more reliable predictions, although there is no database to confirm these conclusions. The primary sources of error is similar to those for the NNP (above), except that estimating long-term reactive sulphide from a short-term test may result in some uncertainty.

In the event the samples from a geologic unit indicate the unit has or may have the potential for net acid generation, kinetic tests should be conducted (Figure 4.4-1, 4.4-2). These tests are discussed in Section 4.5.

4.5 KINETIC TESTS

4.5.1 Introduction

The objective of static teste (Section 4.4) is to identify the geologic units at a site that may have the potential to generate net acidity. Geochemical kinetic tests involve weathering under laboratory controlled or on-site conditions samples of these units in order to confirm the potential to generate net acidity, determine the rates of acid generation, sulphide oxidation, neutralization, and metal depletion and to tost control/treatment techniques. This information is critical because, for example, the rate of acid generation may be negligible, or in extremely rare cases, may be severe for only a short period of time so that long-term control or treatment techniques may not be necessary. Based on the results of kinetic tests the optimization of treatment and control techniques to address the specific severity and duration of acid drainage from a geologic unit will minimize overall costs of acid generation abatement.

Whereas static tests provide some information en overall potential acid generation independent of time, kinetic tests explicitly define reaction rates through time under specific conditions. As a result, kinetic tests are significantly more expensive and continue for months or years. Laboratory kinetic tests conducted in the short term only provide semi-quantitative information on drainage water quality because they de not reproduce site conditions. In order to provide quantitative data on water quality at the site, waste material test pads can be monitored for several years. Ultimately, true prediction of long term drainage quality will only be possible through quantitative mathematical models which can reliably extrapolate results beyond the time of the tests (Section 2.6).

4.5.2 Procedures for conducting a kinetic test

The approach to the kinetic test program is depicted in Figure 4.5-1 and is described in detail below.

The initial step in a kinetic test is the definition of additional material characteristics to those measured in static tests, specifically surface area, mineralogy and total metals. These characteristics are important to the interpretation of the results from kinetic testwork as they can affect the acid generation process or overall water quality.

The particle size of a material (Appendix 4.5, Section 4.5.1.1) can affect the acid production and acid consumption results. Smaller grain-sized materials have a greater surface area per unit weight and a greater density of broken crystal bonds (Lappakko, 1987). Based on experiments conducted in British Columbia, differences in surface area have accounted for apparent differences in rates of different rock types at a proposed mine site (City Resources, 1988).

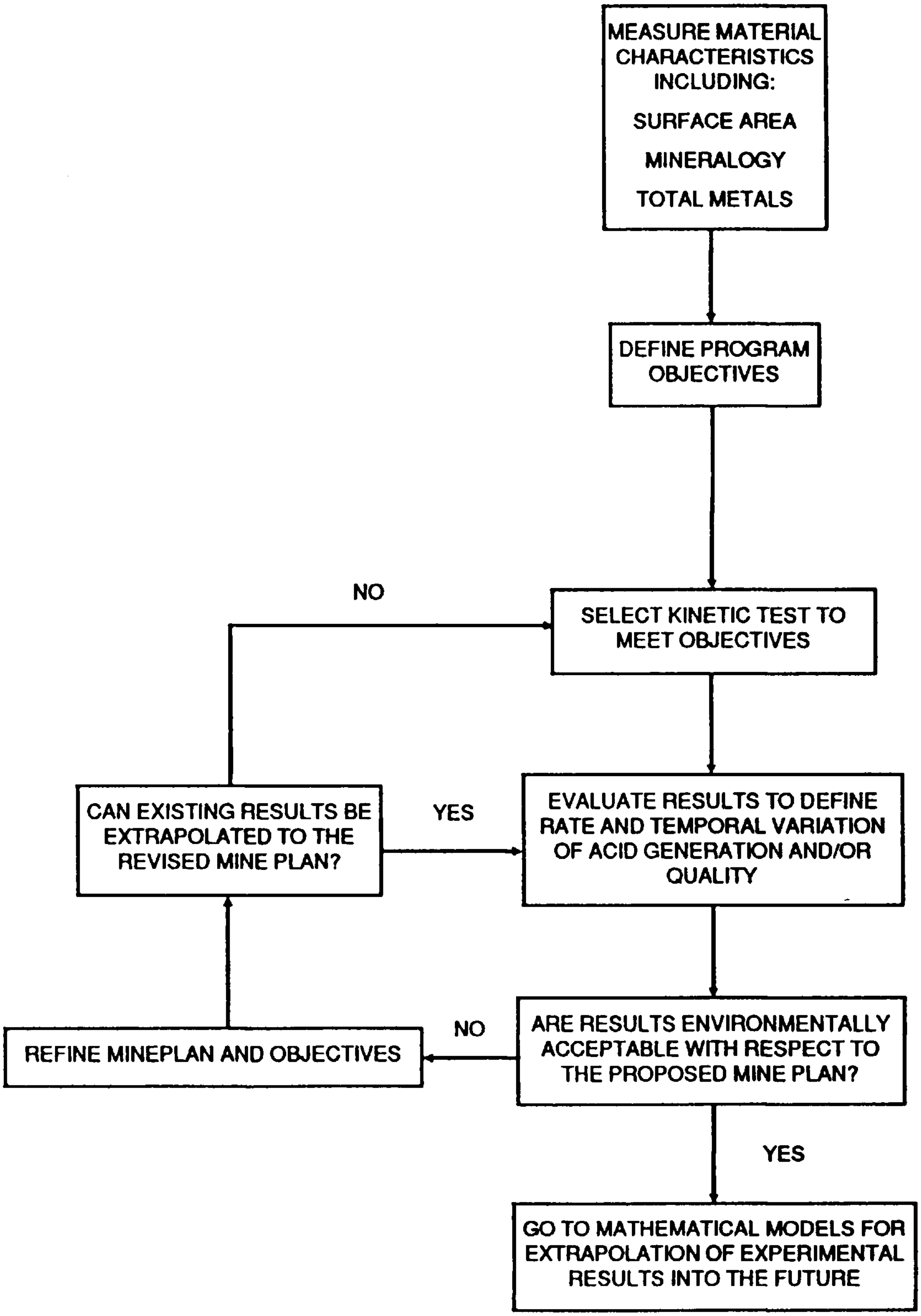


FIGURE 4.5-1
RECOMMENDED KINETIC TEST PROCEDURE

The mineralogy of a sample (Appendix 4.5, Section 4.5.1.2) may also be directly related to reaction rates. Both the chemistry and crystal form of the minerals in a sample controls the rates of acid generation and neutralization. For example, poorly crystalline minerals react faster then their crystalline counterparts and some sulphide minerals oxidize faster than others. Additionally, the mineralogy of a sample may determine the metals that could be leached during acid generation and the extent to which pH may be neutralized by the sample.

Total metal analysis (Appendix 4.5, Section 4.5.1.3) assists in the evaluation of the water quality from the tests. First, total metal analysis indicates any metals present in high levels that may warrant attention. Second, the leaching rates of a metal when compared to the total metal content will suggest when a metal may be depleted within the sample resulting in negligible leach concentrations even though it is difficult to extrapolate laboratory testwork concentrations to field leaching conditions.

Once the material characteristics have been determined which include the sulphur and carbonate content determined during static testwork, the overall program objectives must be defined before selection of a kinetic. test. The program objectives should be based on the mine plan and the proposed handling of acid-generating rock. Program objectives could include one or more of the following objectives:

1. Selection or confirmation of disposal options;
2. Determination of the overall water quality impacts;
3. Determination of the effect of the flushing rates through a sample on water quality; and
4. Determination of the influence of bacteria on the acid generation sample.

Kinetic tests are selected for each acid-generating component based on the information required to meet program objectives. In Appendix 4.5, Section 4.5.2, there are descriptions, procedures for interpretation of the data, and the advantages and disadvantages of each test that is summarized in Figure 4.5-2.

Both small-scale controlled tests (for example, humidity cells) and large-scale on-site weathering trials have been used in assessing acid generation reactions. The controlled tests have the advantage of simulating specific climatic and weathering conditions. On-site tests may be considered more representative than controlled tests because of the natural conditions under which the tests are conducted; however, since results vary as climatic conditions change, the interpretation and extrapolation of the test results is more complicated.

The data from kinetic tests are evaluated to define the rate and temporal variation of acid generation and water quality of a sample or a treatment/control technique. The results are assessed to determine if they are environmentally acceptable with respect to the proposed mine plan. For example, if the proposal is made to mix waste rock with limestone and the testwork indicated that there was acidic drainage from a kinetic test, then the results would not be environmentally acceptable.

TEST	B.C. Confirmatory Test	Shake Flasks	Humidity Cells	Soxhlet Reactor	Columns/lysimeter	Test Plots/Piles
OBJECTIVES						
Selection or Confirmation of Disposal Options			●		●	●
Determination of Overall Quality Impacts		●	●	●	●	●
Determination of Effect of Flushing Rates			●		●	●
Determination of Influence of Bacteria		●	●		●	●
Confirm Potential to Generate Acid under Test Conditions	●	●	●	●	●	●
Determination of Rate and Variability in Rate of Acid Generation		●	●		●	●
ADVANTAGES						
Simple to Use	●	●	●	●		
Test Widely Accepted in Canada			●		●	●
Data May be Used for Mathematical Models		●	●		●	●
Large Numbers of Samples Can be Tested in a Relatively Short Time	●			●		
DISADVANTAGES						
Complex Interpretation		●	●	●	●	●
Long Time to Complete Test		●	●		●	●

FIGURE 4.5-2
COMPARISON OF AVAILABLE KINETIC TEST METHODS

If the results are not environmentally acceptable, then the mine plan and the program objectives must be redefined. The mine plan must be redefined to ensure that the appropriate acid generation control and treatment techniques are used. The program objectives may have to be redefined to incorporate the changes to the mine plan and to test for additional appropriate acid generation control and treatment techniques.

Additional testwork would not be necessary if the existing data is sufficient for extrapolation to evaluate the results as to whether they are environmentally acceptable with respect to the new mine plan. If the existing data is not sufficient, then new kinetic tests should be conducted to meet the new objectives.

When the results are environmentally acceptable, experimental results can be extrapolated to other conditions or into the future using mathematical models. The models that can be used to extrapolate results of kinetic tests are discussed in Section 4.6.

4.6 MATHEMATICAL MODELS

The static and kinetic predictive tests (Sections 4.4 and 4.5) for a proposed mine site provide a significant database of geochemical and water-flow information. This information may be used in mathematical modelling so that it can be consistent with theory and applicable to the proposed mine plan. For example, if a mine plan calls for underwater storage of acid-generating waste rock, data must be presented or extrapolated to demonstrate the environmental impacts of the proposed underwater storage plan. An inherent weakness of predictive testwork is the required extrapolation to predict impacts after years of mine operation from relatively short-term tests. This is particularly critical where acid-generating rock has significant levels of both solid-phase sulphide and solid-phase carbonate so that drainage from the rock may remain pH-neutral for many years before becoming acidic.

The extrapolation of rates of acid generation and any pH neutralization beyond the time frame of the tests can be accomplished using basic mathematical models. Kinetic tests of acid-generating rock demonstrate the existence of temporal variations in acid generation that include one or more of the following trends: (1) an initial rapid release of any acidity and acid products stored in the sample prior to testing, (2) a lag time during which the rate of acid generation accelerates, (3) a peak in the rate of acid generation, and (4) a decline in the rate of acid generation through time. Any neutralization of pH occurring in the sample may follow similar or independent trends. Computer software is commercially available to define equations for best-fit lines. These equations will assist in extrapolating regular, well-defined trends in acid generation and neutralization from short-term predictive tests to years of mine operation and closure.

Predictive test results that indicate complex acid generation and neutralization reactions which follow irregular trends may be predicted for the longer term using a larger system model. A variety of larger system models are described in Chapter 5 which address the complexities of the reactions and can provide detailed simulations of acid generation and neutralization at the site.

4.7 SUMMARY

The potential for waste to generate acid rock drainage may be evaluated using several literature, laboratory and field based approaches.

As an initial approach, the proposed mine can be compared with other mines that are in similar geological, paleoenvironmental and geographical settings. This approach is extremely limited, particularly in the case of metal mines where similar geological environments may not exist elsewhere.

Static predictive tests allow determination of the potential to generate net acidity by analyses for sulphur, in its various forms, and neutralizing minerals. Because these tests do not determine the release of acidity as a function of factors such as mineralogy and time they can not be used to quantitatively predict water quality. However, the tests do provide a qualitative indiction of whether or not net acidity will be produced. Sufficient samples should be tested to adequately represent heterogeneities in the rock which will affect the potential for acid generation.

Kinetic tests are used to confirm the results of static tests and determine the rate of acid generation and neutralization. Testing involves leaching of representative samples and monitoring of water quality over a period of months or years under laboratory or field conditions. Laboratory testing allows greatest control of factors which affect acid generation but can not be used to predict water quality under field conditions. Field kinetic tests provide a more reliable indication of rates and duration of acid generation under site conditions.

Finally, mathematical models are used to extrapolate static and kinetic test data to long term prediction of water quality. This can be achieved either by determining best fit curves which can be reliably extended beyond the time range of the data, or through large system models which theoretically attempt to model acid generation processes.

CHAPTER 5: ACID GENERATION AND METAL LEACHING MODELS

5.1 INTRODUCTION

The extrapolation of rates of acid generation and possible pH neutralization beyond the time frame of laboratory and field tests can be attempted using simple mathematical expressions, or by using more complex mathematical models incorporating a number of inter-related mathematical functions. The more basic models aim to predict acid generation or water quality by extrapolating trends in data obtained from short-term testing, to periods of years or tens of years. These models are generally equations of best-fit lines applied to laboratory or field test data. A number of equations for best-fit lines are available, the most suitable of which, for a particular application, would depend on the data. The more complex models attempt to predict acid generation and metal leaching by simulating the different, interactive processes by means of mathematical expressions. This chapter of the Guide addresses development of these more complex models.

A discussion on the development and application of mathematical models in the field of ARD prediction, prepared by SENES Consultants Ltd., was presented in a report on metal leaching from mine wastes for the American Mining Congress (Steffen Robertson and Kirsten, 1988). Mathematical models may have application in the long-term prediction of ARD in British Columbia. This chapter is based on the discussion presented by SENES Consultants Ltd., referenced above.

Mathematical models have been developed that represent some key oxidative, redox, or electrochemical processes that cause the dissolution of minerals. However, by their nature, models can be only a partial representation of these natural processes and contain a number of parameters that are either imperfectly known or inherently variable in nature. For example, the rate of biological oxidation of pyrite is known to depend on, among other variables, the degree of crystallinity and the particular variant of Thiobacillus ferroxidans species. In this case, the mineralogical and the biological parameters are often not well known and defy precise quantitative description.

In general, three sources of uncertainty in model predictions arise:

- Incomplete or invalid model structure,
- inherent parameter bias or variability,
- lack of parameter calibration and model verification.

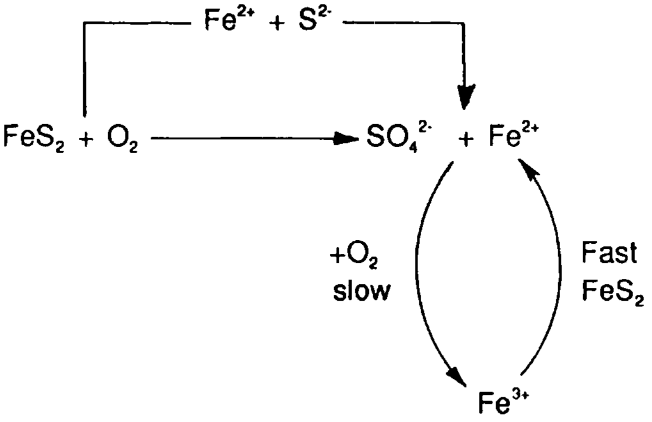
Incomplete or invalid model structure results from our poor understanding of the rate limiting processes or from the overuse of simplifying assumptions. An example would be the application of an equilibrium approximation for a kinetic process. Since models are only partial representations of the real system, a variety of alternative models of varying complexity can be derived for a leaching system. It is often difficult to determine which (if any) of the models is the most appropriate representation of reality. Increasing model complexity does not necessarily lead to better predictive ability, as more complex models require more model parameters, which themselves must be determined. Often, model parameters are not constant, but possess some statistical distribution (particle size, sulphur content, rainfall, for example) which may have a significant effect on model predictions. Often, the most important shortcoming is the inability of the model developer to calibrate the model parameters and verify the model. This arises from the lack of sufficient data in both a spatial (depth of tailings, for example) or in a temporal sense (too short sampling periods). In view of these, the uncertainties of model predictions, particularly long-term predictions, must be recognized by the model builder as well as the model user.

Traditionally, a deterministic approach has been used in constructing acid generation and metal leaching models. Deterministic models result from considering the mechanism underlying the process and assign some values to the significant parameters. These parameter values are often based on some laboratory leaching tests. The tendency in using deterministic models has been to use conservative assumptions about the parameters to ensure that the environmental releases are not underestimated. The use of laboratory derived data generally does result in conservative predictions with respect to acid generation and metal leaching, since the tendency of the experimenters is to create ideal conditions that are not achieved in nature. "Overconservatism", however, is a pitfall, that a model user may face. This may distort one's perception of reality and complicate rather than simplify decision making (SENES, 1984). Overconservatism may become a real problem when the predicted effects are perceived to be "significant" by either those affected or the regulatory authority or both.

5.2 MECHANISMS OR PROCESSES SIMULATED

Any chemical or biochemical mechanism consists of a series of elementary processes. In each elementary process both the molecularity and reaction order are preserved. The slowest of these elementary processes is regarded as the rate determining step.

Acid production from tailings and waste rock involve, almost invariably, reactions of metal sulphides. Of these mineral metal sulphides, the iron minerals (pyrites, marcasite, pyrrhotite) are the most abundant and, therefore, the most significant. Leaching processes involving the more "glamorous" sulphide minerals have been the subject of detailed studies both from a mechanistic and economic viewpoint. A large portion of this information describes the thermodynamics and the kinetics of the sulphide leaching reactions. It has been pointed out by Lowson (1982), that, in contrast to other metal sulphides, the mechanism of the aqueous oxidation of pyritic minerals is imperfectly known in spite of their abundance. Stumm and Morgan (1981) proposed the following reaction sequence for the oxidation of pyrite:



(5.1)

The above diagram is a simplification of the chemical reactions (2.1), (2.2) and (2.4) presented in Chapter 2.0.

Metal sulphide oxidation in general, pyrite oxidation in particular, may involve inorganic processes, microbial mechanisms, or both. Mechanistic studies are often aimed to determine which of these processes are the most important controlling the rate of leaching. The relative significance may affect the "at source" control of the acid generation and metal leaching.

5.3 MODEL SELECTION AND DEVELOPMENT

The objective in model selection and development is to derive mathematical expressions that are good conceptual representations of reality. Deterministic geochemical models for metal leachings consist of dynamic (time dependent) and equilibrium (independent of time) and empirical expressions. The use of these expressions as model components for acid generation and metal leaching are summarized in Table 5.3-1. Metal sulphide and ferrous ion oxidation, heat generation and transport, diffusive and convective transport of mobile reactants (oxygen, for example) are regarded as dynamic processes. These processes are expressed as a series of differential equations. On the other hand, non-oxidative dissolution, aqueous speciation, neutralization reactions, solute adsorption and ion exchange are usually modelled as chemical equilibria. In this case, mathematical representations are in terms of algebraic equations that require simultaneous solutions. Empirical algebraic equations have been applied to simulate seasonal variables (atmospheric precipitation, ambient temperature variations) or for parameter estimation (temperature and pH effects on reaction rate constants, for example).

TABLE 5.3-1

MATHEMATICAL EXPRESSIONS IN GEOCHEMICAL MODELS

(After Steffen Robertson and Kirsten, 1988)

TYPE OF EXPRESSION	PROCESS SIMULATED
Dynamic	. Oxidation kinetics . Heat (energy) transport . Transport of oxygen
Equilibrium	. Solids dissolution . Aqueous speciation . Neutralization/buffering . Ion exchange . Surface adsorption/complexation
Empirical	. Periodic (seasonal) variation of temperature, atmospheric precipitation . Parameter estimation

The primary output variable at most acid generation models has been the rate of sulphate generation from pyrite. This rate can be stoichiometrically related to the rate of pyrite and oxygen consumption by the following equalities:

$$\frac{1}{2} \quad \frac{d\{SO_{42-}\}}{dt} = \frac{-d\{FeS_2\}}{dt} = \frac{-1}{3.5} \quad \frac{d\{O_2\}}{dt} \tag{5.2}$$

where:

$\{SO_4^{2-}\}$ = mass of sulphate (moles)
$\{FeS_2\}$ = mass of pyrite (moles)
$\{O_2\}$ = mass of oxygen (moles)

The important developments in modelling waste rock and tailings are summarized in Table 5.3-2. Mathematical modelling of waste rock dumps began during the last decade. In these early model developments, it has been correctly recognized that the transport of gaseous oxygen to the reaction site plays a critical, rate-limiting role with respect to acid generation (sulphate oxidation).

TABLE 5.3-2
ACID GENERATION/METAL LEACHING MODELS FOR WASTE ROCK DUMPS AND TAILINGS

(After Steffen Robertson and Kirsten, 1988)

Application	Model Characteristics	Reference
aste Rock Dump	. Oxygen diffusion through pores . Undifferentiated surface reaction model	Morth and Smith 1966
Waste Rock Dump	. Oxygen diffusion through porous slab . Simple homogeneous reaction model	Ritchie, 1977
Waste Rock Dump	. Convective oxygen transport through pores . Oxygen and heat balance . Shrinking reactive core model	Cathles and Apps, 1975
Waste Rock Dump	. Two dimensional convective oxygen . Biological and inorganic oxidations . Oxygen and heat balance . Shrinking reactive core model	Cathes, 1979 Cathles and Schlitt, 1980
Waste Rock Dump	. Serial diffusion of oxygen through pore space and within particle to reaction site . Multi-layered configuration . Biological and inorganic oxidations . Simple chemical speciation . Shrinking reactive core model	Jaynes et al., 1984
Waste Rock Dump	. Serial diffusion of oxygen through pore space and within particle . Heat source distribution . Numerical and approximate analytical solutions . Shrinking reactive core model	Davis and Ritchie, 1982, 1986
Tailings	. Diffusion of oxygen through pore space, variation with moisture content . Biological and inorganic reactions . Temperature, pH, nutrient effects . Chemical speciation . Particle size distribution . Shrinking radius model	SENES and Beak, 1986

In more recent model developments (Cathles and Schlitt, 1980; Jaynes et al., 1984; Davis and Ritchie, 1986) two important oxygen transport processes have been recognized in waste rock dumps. For modelling purposes, waste rock is regarded as spherical particles in which pyrite and/or other oxidizable sulphide minerals are more or less uniformly distributed as a network of inter-connected grains. Oxygen is transported in the pore space between the particles. In very coarse waste rock, the oxygen transport in this pore space appears to be by convection (Cathles and Apps, 1975; Cathles, 1979; Cathles and Schlitt, 1980), while in finer waste rock (Davis and Ritchie, 1986) and coal strip mine refuse (Jaynes et al., 1984) the oxygen transport in this pore space has been assumed to be diffusive. The mass balance of oxygen in the pore space has been described as follows:

$$\text{Øa}.\ \frac{\partial C}{\partial t} = D_e \frac{\partial^2 c}{\partial z^2} + q \qquad (5.3)$$

where:

Øa = air filled porosity (m^3m^{-3})
C = concentration of oxygen in pore space (moles m^{-3})
t = time(s)
D_e = effective diffusivity of oxygen in pore space (m^2s^{-1})
q = oxygen consumption rate (moles O_2 m^{-3} s^{-1})

The concentration of oxygen given by Equation (5.3) refers to the surface of the particle. The concentration from the particle surface to the moving reaction front within the particle is given by:

$$D_e^* \quad \frac{\partial^2 C^*}{\partial r^2} + \frac{2}{r}\frac{\partial C^*}{\partial r} = -k\ a\ C_s^* \qquad (5.4)$$

where:

C* = concentration of oxygen within the particle (moles m^{-3})
D_e^* = effective diffusion coefficient of oxygen within the particle (m^2 s^{-1})
r = radial distance within the particle (m)
k = first order reaction rate constant per unit surface area (s^{-1} m^{-2})
a = surface area of the reacting front (m^2)
C_s^* = concentration of oxygen at the reacting front (moles m^{-3})

Equation (5.4) is known as the shrinking reactive core model. If X is defined as the fraction of unreacted pyrite at any time, t, a rather simple expression for the rate of change of unreacted pyrite has been derived by Jaynes et al. (1984) as follows:

$$\frac{dX}{dt} = \frac{-1}{2\,t_D\,(1-X) + t_c} \qquad (5.5)$$

where:

X = fraction of unreacted pyrite within the particle
t_D, t_c = characteristic time constants (s)

The time constant, t_D, has been defined by Levenspiel (1972) as the time required to oxidize all the pyrite within the particle when the diffusion of reactants is the rate limiting step. This total time, t_D, has been related to particle parameters in the following manner:

$$t_D = \frac{e_p\,R^2}{2\,b\,D_e\,C} \qquad (5.6)$$

where:

e_p = molar density of pyrite (moles m^{-3})
R = particle radius
b = stoichiometric ratio of pyrite to oxidant consumption
D_e = effective diffusion coefficient of oxygen within the particle ($m^2\ s^{-1}$)

The second time constant, t_c, has been defined as the total time requirement for pyrite oxidation if the chemical oxidation on the surface is the rate determining step:

$$t_c = \frac{e_p.R}{b\,K_s\,C_o.d.a} \qquad (5.7)$$

where:

K_s = first order surface reaction rate constant (m s^{-}1)
d = effective thickness of the pyrite oxidizing zone (m)
a = pyrite surface area per unit volume ($m^2\ m^{-3}$)

As implied by Equation (5.5), the rate of pyrite oxidation is "reaction controlled" initially and "diffusion controlled" on a long-term basis in waste rock dumps. In tailings consisting of finely ground waste rock of less than 500 um diameter, however, pyrite tends to occur as distinct particles. In this case, oxidation occurs along the entire particle surface. It has been shown that this oxidation results in the shrinkage of the entire particle, hence the rate of pyrite oxidation, is kinetically controlled. SENES and Beak (1986) has developed a mathematical model for specific application with fine tailings. The model, known as the RATAP model, comprises several modules: initial inventory; kinetics; oxygen transport; pyrite oxidation; solute transport; and aqueous speciation. In the initial inventory module, values for parameters that are either periodic (soil temperatures) or constant (oxygen partial pressure in air) are calculated. The kinetics module determines the biological and chemical rates of pyrite oxidation per unit pyrite surface area as a function of temperature, pH, moisture content, oxygen concentration, carbon dioxide, and phosphate levels. Pyrite particles are expressed by a size distribution density function. The oxygen concentration with depth is calculated by the oxygen transport module. The tailings area is divided into 20 vertical layers of varying moisture and pyrite content. The oxidation of pyrite in each layer is calculated as a function of the oxygen concentration. Lastly, the solute transport and aqueous speciation models are used to calculate aqueous and solid phase concentrations of various dissolved species, the tailings porewater pH, and the acid flux leaving the tailings.

Equilibrium geochemical models are non-specific for waste rock or tailings, nevertheless, they are often useful in evaluating the solution of minerals in groundwater below some oxidizing tailings or waste rock pile. Comprehensive computer models for calculating aqueous equilibria include the MINEQL (Westall et al., 1976), the WATEQ3 (Ball et al., 1981) and the MINTEQ (Felmy et al., 1984). The MINTEQ model is perhaps the most advanced. Besides evaluating solubility and aqueous speciation, this model is capable of calculating ion exchange and surface adsorption by "activity Kd", "activity Langmuir", and "activity Freundlich" isotherms. Equilibrium constants are calculated from thermodynamic data and corrected for temperature. Activity coefficients for ionic species are calculated by the Davis equation. The equilibrium expressions are believed to be applicable to an ionic strength of approximately 0.7 molal. However, the use of the equilibrium expressions can be extended to higher ionic strengths by considering higher order ion interaction models such as described by Pitzer (1973), Pitzer and Kim (1974).

5.4 INPUT DATA REQUIREMENT

The input data requirement depends on the complexity of the model. The total number of input parameters may range from approximately 22 for simple models for dump leaching with oxygen and heat balance (Cathles and Apps, 1975) to 73 input parameters for multi-layered models with speciation (SENES and BEAK, 1986). In general, input parameters include; a) the inventory of waste rock/tailings; b) kinetic parameters; c) oxygen transport parameters, and; d) solute transport and speciation parameters.

Waste rock/tailings inventory includes bulk density, particle size distribution, mass fraction of pyrite and other solids, porosity, and moisture content. The kinetic parameters include the biological and chemical surface oxidation rates as a function of temperature, oxygen concentration, pH, etc. The oxygen transport parameters consist of the relevant diffusivity coefficients in the airfilled pore space, within the particle (shrinking reactive core models), and convective oxygen transport parameters.

Solute transport and speciation involves the calculation of annual precipitation, solids dissolution, and aqueous speciation to calculate the dissolved species concentration and the pH. In many cases, the parameters are imperfectly known. This is particularly true for parameters involving biological oxidation and oxygen transport. The determination of parameters associated with these processes though a series of laboratory studies with the specific waste material is necessary in most cases.

5.5 MODEL APPLICATIONS

To be used as predictive tools, mathematical models require verification, calibration, and validation. Verification involves the examination of the fundamental equations and the underlying assumptions for correctness. The model structure might be reviewed by independent experts. The next step is to check the computer code. Computer-based models often contain lengthy, often complex algorithms. It is difficult to generate error-free codes, therefore, structured (modular) programming that is easy to follow is recommend. An aspect that is often neglected is the robustness of the model. A robust model is one which produces consistent results throughout the entire possible range of input parameters. This is often evident in case of chemical models that attempt to simulate water quality (pH, for example) during both "high" and "low" sulphide oxidation activity. The robustness can be assessed by parameter sensitivity analysis.

Model calibration means the determination of the "best" parameter values. The measure of the "goodness of fit" is some statistical property, such as the residual sum of squares between observations and predictions. Historical records from existing tailings or waste dumps are the best for this purpose, however, the database is seldom available in sufficient detail to give any credence of the method. Long-term field leaching studies with test stockpiles (Eger and Lapakko; 1985; Lapakko, 1987) are particularly useful, since they allow the assessment of the effect of time on the various environmental processes. If no field data is available, laboratory tests should be performed. These tests may mimic natural processes by artificial acceleration of the time dependence. Silver and Ritcey (1980) used lysimeters filled with tailings and simulated sunlight, darkness, precipitation, etc., to accelerate nine-fold the natural oxidation process. If no site-specific tests are available, the modeller should resort to published data from other sites to establish a range of parameter values. However, data derived from different sites must be used with caution, since oxidation of pyrite, particularly bacterial oxidation, may vary more than one order of magnitude at even similar pyrite content, temperature, and pH.

Validation means the application of a calibrated model to a new set of data collected at a different, but similar location or at the same location at a different time. Due to the general lack of long-term field data, it is unlikely that any model developed for acid generation in waste rock dumps or in tailings is fully validated.

Cathles (1979) has applied a model based on shrinking core reaction kinetics for the industrial leaching of copper from low grade sulphide ores. The model was used to predict the time course of copper extraction (% copper leached). The authors (Cathles and Apps, 1975) used temperature and oxygen profiles for parameter calibration at five dumps of the Utah Copper Division of Kennecott Copper Corporation. The model was then partially validated using the temperature and leaching data of the thermally insulated test tank at the John D. Sullivan Center for In Situ Mining, New Mexico Institute of Mining and Technology, Soccoro, New Mexico. The model was capable of predicting general trends, but consistently overpredicted the temperature in the test column. It should be noted that due to the highly exothermic nature of the pyrite oxidation reaction, the temperature rise resulting from the enthalpy change of the reaction is a convenient means of measuring the extent of the reaction.

A similar model based en shrinking reactive particle core was developed by Jaynes et al. (1984) for predicting pyrite oxidation and effluent water quality in reclaimed coal strip mines. In contrast to Cathles' (1979) treatment, the authors considered diffusive transport of oxygen in inter-particle pore space and introduced a separate reaction term for bacterial iron oxidation (indirect leaching). The model was calibrated by the extensive use of published data and the model was tested extensively by simulating field data. The model, however, has not been validated.

The mathematical model development by Ritchie (Ritchie, 1977; Harries and Ritchie, 1981; Harries and Ritchie, 1983; Davis and Ritchie, 1982, 1986; Davis, Doherty and Ritchie, 1986) illustrates the evolution of modelling from a homogeneous, porous slab to a model involving double diffusive transport of oxygen coupled with a shrinking core model. The authors have used temperature and oxygen concentration profiles for model calibration and partial validation. The authors compared the results of numerical solutions with approximate analytical solutions that are easier to apply. Although the approximate solutions are believed to be less accurate in predicting oxygen and temperature profiles, they are reported to provide an insight into the importance of the various mechanisms.

A comprehensive model, known as the RATAP model, has been developed by SENES and Beak (1986) for modelling acid generation and ground-water quality in fine pyritic tailings. This model incorporates the concept of size distribution and shrinking particle size. Separate kinetic modules for bacterial and inorganic pyrite oxidation have been developed. The model has undergone extensive verification (sensitivity analysis) and parameter calibration using in-situ oxygen, sulphate, calcium, iron, and aluminum concentrations and pH profiles. The model has been partially validated by the application of model predictions to a monitored uranium tailings site in Northern Ontario.

The above mentioned models represent the most concerted approach to describe the sulphide oxidation mechanism, the transport processes, and predict acid generation in waste rock and tailings. Although other models are also available, these are scarcely more than hydraulic flow, chemical mixing, or generic geochemical equilibrium considerations. It is worth noting that the common feature of the mechanistic models is the importance of the transport of gaseous oxygen through the solid matrix. The models differ only in a few assumptions concerning the transport and the combination of inorganic and biological reaction phenomena.

All models suffer from the lack of sufficient field data for complete validation. True model improvement can be realized only when a more extensive database becomes available.

5.6 CONCLUSIONS

1) Kinetic test results provide a basis for predicting the rate of acid generation that is likely to occur in the field over periods similar to that of the test duration. Mathematical models are used to extrapolate the results of prediction tests over periods of many years, decades, or even centuries. Mathematical models may range in complexity from a single, relatively simple, mathematical expression that describes a best-fit extrapolation of available data, to detailed, complex models that simulate a number of interactive processes.

2) A number of different models have been developed for application to waste rock and tailings, as described in Section 5.3. The complexity of the model selected must be compatible with the level of available input data. Models may be developed to suit site specific needs or existing models may be used if a suitable and applicable model is available.

3) Mathematical models require verification, calibration, and validation before these can be confidently used as prediction tools. Care should be exercised in selecting models in order to choose a model that suits the need of the application, and is compatible with the sophistication of the input data. There is uncertainty with the results of model predictions because, at present, all models suffer from a lack of field data for complete validation.

4) In many cases, the more simple mathematical models that can be used to extrapolate trends in laboratory or field test data are more useful than complex models due to lack of input data and uncertainty in the complex models.

CHAPTER 6: CONTROL OF ACID GENERATION

6.1 INTRODUCTION

Methods of controlling acid generation include prevention and abatement techniques. Prevention refers to measures designed before mining starts and with the knowledge of the acid generation potential of the waste. Abatement refers to measures implemented either at facilities where ARD is occurring and was not anticipated, or at facilities where control measures are not sufficiently effective.

Much of the effort in controlling acid rock drainage in the past has been directed at treatment of the leachates, i.e., addressing the symptoms rather than the cause. Until quite recently, most treatment processes involved neutralization techniques without attempting to control the problem at the source. This is possibly due to the fact that the problem was in many cases not anticipated and only became evident once acid drainage was being produced. Further, the processes involved in the generation of acid drainage have only recently become better understood and are, as yet, not completely understood. A sound understanding of the processes and conditions is important in the design of an appropriate control measure.

The importance, objectives and approach to ARD control are discussed in Chapter 1.0. The three stages of ARD control are:

1) Control of the acid generation process.
2) Control of acid drainage migration.
3) Collection and treatment of acid rock drainage.

The control of ARD by preventing or inhibiting acid generation is the most preferable level of control. If acid generatien is prevented there is no risk of the rosultant contaminants entering the environment. This chapter of the guide describes the objectives and methods for control of acid generation. The control of ARD migration and ARD collection and treatment are discussed in Chapters 7.0 and 8.0, respectively while Chapter 10.0 contains recommendations for ARD control.

6.2 OBJECTIVES OF ACID GENERATION CONTROL

The objective of acid generation control is to prevent or reduce the rate of acid the formation at the source by inhibiting sulphide oxidation. This may be done by excluding one or more of the principal ingredients or by controlling the environment around the sulphides. The process of acid generation and associated chemical reactions are described in Chapter 2.0.

The primary ingredients in the acid generation process are firstly; wastes containing reative sulphide, secondly; oxygen and thirdly; water. For practical purposses the first stage of the chemical reaction would not proceed and no acid production would result without these components. Factors that influence acid generation, in terms of rate of production, Include bacterial activity, temperature and pH.

6.3 AN APPROACH TO ACID GENERATION CONTROL

Acid generation can be controlled by eliminating or reducing one or more of the essential components, or by controlling the environment at the source in order to retard the rate of acid generation to an insignificant level. The approach to controlling acid generation by these means is discussed in this section. The control of acid generation requires one or more of the following:

i) Sulphide removal or isolation

If the sulphide minerals in tailings and waste rook are removed, reduced to insignificant levels, or isolated by a coating or some other means, the chemical reactions that produce acid through sulphide oxidation can not occur. It may be possible to concentrate high sulphide content wastes and to separate these from the bulk of the mine waste. Procedures that concentrate the highly acid generating wastes may have definite potential in waste management aspects. However, some form of ARD control is nevertheless required for the high sulphide waste, either treatment, removal from the site or in-situ control. The procedures that concentrate, remove, or isolate the sulphides are referred to as "conditioning of tailings and waste rock" in this guide and are discussed in Section 6.5.

ii) Exclusion of water

Exclusion of water to the extent that acid generation does not occur is not considered practical (Robertson, 1987). This would require the exclusion of surface water, infiltration due to precipitation and groundwater seepage. The main source of water depends on the type and location of the waste facility. For example, groundwater seepage is likely to be prominent in underground mines while surface water and infiltration may be critical for waste rock dumps and tailings deposits. The exclusion of water can only be achieved with impermeable barriers such as a synthetic membrane cover. It is generally accepted that long-term degradational effects would result in sufficient water penetrating the barrier to enable acid generation to proceed.

iii) Exclusion of oxygen

Eliminating oxygen from reactive wastes would prevent the primary chemical reactions responsible for acid generation from occurring. Reducing oxygen access may be of benefit in retarding the rate of production of contaminants. Although it is chemically possible to generate acid under anaerebic conditions by, for example, reation of sulphur species in chloride solutions, this has not been shown to be a significant factor in mining wastes.

The exclusion of oxygen to the extent that acid generation is reduced to acceptably low levels generally requires the placement of a cover with an extremely low oxygen diffusion characteristic. The exclusion of oxygen from reactive wastes is probably the most effective long-term acid generation control technique. Appropriate cover materials for the purpose of reducing oxygen access include water, soil, synthetic materials and combinations of these materials. Covers and seals to exclude oxygen are discussed in Sections 6.9 and 6.10 of the Guide.

iv) Temperature control

If permanently frozen conditions can be achieved, acid generation can be prevented. This control measure may have application In regions of permafrost, with adequate precaution being taken to prevent seasonal thawing.

v) pH control

If the pH of interstitial water in waste is maintained within the alkaline range, acid generation production as a result of sulphide oxidation will be inhibited. The pH may be controlled by the addition of alkaline materials to potentially acid generating wastes. This may be achieved either by blending net acid consuming wastes with the acid producing waste to achieve a net acid consuming mixture, or by adding and mixing imported alkaline material such as ground limestone. The techniques of waste segregation and blending, and base addition are discussed in Sections 6.6 and 6.8, respectively.

vi) Control of bacterial action

Once the pH within a reactive waste pile drops below about 4 bacterial oxidation, principally by Thiobacillus ferrooxidans, Increases the rate of acid generation by five fold or more. Anionic surfactants (in particular sodium lauryl sulfate), organic acids and food preservatives have been used as antibacterial compounds to control bacterial action. The use of bactericides to control acid generation is discussed In Section 6.7.

6.4 AVAILABLE CONTROL MEASURES

The objective in the design of ARD control is to achieve the necessary control to satisfy environmental criteria using the most cost effective technique. The suitability and effectiveness of any control measure is dependent on a number of site specific conditions. Criteria that need to be considered in the selection of control measures include:

1) The degree of acid producing potential of the mine waste. This includes the nature, quantity and reactivity of sulphide minerals present, neutralizing potential of the rock, etc., and would be determined using geochemical testing.

2) The type and physical characteristics of the waste.

3) Site criteria such as climate, topography, surface and groundwater hydrology.

4) The time period for which the control measure is required to be effective.

5) The sensitivity of the receiving environment to acid mine drainage.

It may be that certain measures are not by themselves sufficient to control the release of contaminated drainage from a given site. A combination of measures may yield the most cost effective means to achieve the desired control. The control objectives and available measures for control of acid generation are summarized in Table 6.1 below and discussed in the following sections of this chapter.

TABLE 6.1-1

AVAILABLE ACID GENERATION CONTROL MEASURES
(After Steffen Robertson and Kirsten, 1988 (a))

Objective of Control	Control Measure
Sulphide removal or isolation	• Conditioning of tailings/waste rock
Exclusion of water	• Covers and seals
Exclusion of oxygen	• Subaqueous deposition • Covers and seals (other than water)
pH control	• Waste segregation and blending • Base additives
Control of bacterial action	• Bactericides

6.5 CONDITIONING OF TAILINGS/WASTE ROCK

The generation of ARD may be reduced by placing tailings and rock dumps in a condition that is favourable for ARD prevention. The sulphide content of tailings, for example, may be reduced by means of bulk sulphide flotation prior to placement (Broman, 1988; Hester & Associates, 1984). This process will produce a sulphide concentrate and flotation tailings. The latter will contain residual sulphide and hence may still be potentially acid generating, however, control of acid generation will be easier to achieve for this, the bulk of the waste. The disposal of sulphide concentrate remains a consideration. An option that has been identified is pressure leaching of the concentrate to produce acid, filtering the acid off and disposing of the remainder of the concentrate (Hester and Associates, 1984). The cost of flotation and disposal of sulphide concentrate will influence the feasibility of this method. Another approach to tailings disposal may be to utilize the "dry" tailings disposal technique combined with an additive such as cement or bentonite, for example. These procedures could be used to produce a compacted soil cement with the intention of reducing oxygen and water access to the sulphide minerals. There are at present no available records of these methods having been used to control acid generation.

The potential for isolation of pyrites by developing a coating of some form has been evaluated by some workers (Hester and Associates, 1984), however, these methods are still experimental and do not yet indicate adequate, economical control of ARD.

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 a placement method often adopted. This technique is often referred to as 'sub-aerial' (Knight and Haile, 1983) or 'semi-dry' placement. While this technique may have benefits under certain conditions, abatement of ARD does not necessarily occur. Provided the tailings remain in a saturated state, reductions in acid generation due to oxygen exclusion and reduced infiltration (due to reduced surface permeability) are noticeable but still comparatively small. However, once the tailings are allowed to dry (which is the case in this method) shrinkage cracks extending from the surface into the tailings deposit may cause a dramatic increase in permeability. Evidence indicates that this secondary permeability permits oxygen and water entry into the tailings and hence continued acid generation (Steffen Robertson and Kirsten, 1987b). Where underdrainage is maintained this may increase the rate of both oxygen entry and ARD. Thus the direct beneficial effect of layered tailings, on ARD 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 ARD 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.

For acid generating waste rock, there may be merit in segregating and isolating the high sulphide wastes during mining. This may serve to concentrate the high sulphide wastes in one location. While processes that concentrate high sulphide wastes may have definite potential in terms of waste management aspects, some form of ARD control is nevertheless required.

6.6 WASTE SEGREGATION AND BLENDING

Waste segregation involves the careful removal and separate handling of various geologic units at a minesite. Mines with acid-generating geologic units may also have other geologic units with excess acid-consuming capability. As a result, the segregation and separate handling of each unit provides two primary benefits. First, the volume of rock that may generate acidity and require treatment or control is minimized. Second, if acid-consuming units contain carbonate which readily and reliably reacts to acidic-pH conditions, these units can be blended with the acid-generating units in experimentally defined proportions for pH control. This is practiced effectively at some coal mines in eastern U.S.A. (Skousen, et al, 1987; Sturm, 1987) and on Vancouver Island, B. C. (Milner 1987).

The blending of acid-generating and acid-consuming rock units is similar to the alternative control technique of adding limestone or other neutralizing additives to the acid-generating waste (Section 6.8). Consequently, the success of the blending is primarily dependent on the same factors as limestone addition: (1) the movement of water through the system, (2) the nature of contact of acidic waste/water with the acid-consuming rock/water, (3) the proportion of excess acid-consuming rock, and (4) the type and reactivity of the acid-consuming minerals. These factors determine the required procedure for blending and are discussed in Section 6.8.

Because acid-consuming rock unite rarely contain $CaCO_3$ and other highly reactive carbonates in high proportions, overall costs/benefits of transporting and adding low-volume, highly neutralizing additives (Section 6.8) may be less than blending with higher-volume, less reactive acid-consuming rock. The difference in volume of the mixture, associated catchment area, and monitoring requirements are also facters in determining the overall cost/benefit analysis of the alternatives. However, there are situations where a mine plan may require similar handling, transportation, and disposal for both acid-generating and acid-consuming rock and, if blending is experimentally demonstrated to be successful and reliable, the sole cost of blending of the rock units may be more economical.

The costs of segregation and blending are site specific and are dependent on the mine plan, the handling and transportation of material, and the technique of blending.

6.7 BACTERICIDES

The rate of sulphide oxidation and acid generation is enhanced in some environments by microbiologic activity, particularly that of Thiobacillus ferrooxidans. This bacterial activity can accelerate the oxidation both of ferrous iron to ferric iron in water (Sections 2.3 and 2.4) and of reduced sulphur in the sulphides to a higher oxidation state (Table 2.3-1). The purpose of bactericides is to create a toxic environment for bacteria so that the inorganic rate of acid generation can not be enhanced. This does not imply that acid generation will cease.

The most popular bactericides for acid-generating materials include: benzoate compounds, sorbate compounds, anionic surfactants such as sodium lauryl sulphate, and phosphate compounds. Laboratory and field experiments indicate bactericides reduce the rate of acid generation (as indicated by sulphate and acidity) as well as concentrations of certain metals generally by factors of up to five, with one reported case of a 20-fold reduction (Kleinmann and Erickson 1983; Kleinmann 1981; Erickson et al. 1985; Rastogi and Sobek 1986; Watzlaf 1986; Sobek 1987; B.F. Goodrich Co., 1985). The overall effectiveness of each bactericide compound appears to be similar, generally between 50% and 95% effective in the short term. (Table 6.7-1). It should be noted that the period during which bactericides remain effective is limited due to the fact that they degrade and are removed by infiltrating and percolating water. The results quoted in Table 6.7-1 do not take into account degradation and depetion of bactericides.

TABLE 6.7-1

EFFECTIVENESS OF BACTERICIDAL METHODS

Method	Results	Reference
Spray and controlled release pellets	80% reduction in acidity, sulphates, irons, manganese and aluminum	Rastogi and Sobek (1986)
Sodium lauryl sulphate, potassium benzoate, potassium sorbate on silver mine waste rock	Complete inhibition of T. ferro-oxidans. 92 to 84% lower acidity	Watzlaf (1986)
As above on coal mine waste	Short-term reduction in acidity	Watzalf (1986)
Sodium lauryl sulphate controlled release from rubber pellets	50 to 95% reduction in acidity	Kleinmann and Erickson (1983) Erickson et al. (1985)
Sodium lauryl sulphate	60 to 95% reduction in acid production	Kleinmann and Erickson (1983)
BF Goodrich ProMac System Co.	58% to 72% reduction in acidity, 58% to 68% reduction in sulphate	Sobek (1987) BF Goodrich (1985)

Available studies do not discuss the impact of bactericides on pH, presumably because there is little effect on pH or effects are unpredictable. A ten fold decrease in acidity theoretically changes the pH by only 1 pH unit (for example, from pH=2 to pH=3), although aqueous buffering by sulphate and metals would further limit the pH change. The one detailed set of pH data in the referenced studies (Sobek, 1987) indicated successful inhibition of Thiobacillus ferrooxidans resulted in a maximum pH increase of only 0.3 units from pH 2.9 to pH 3.2.

Based on the laboratory and field data, bactericides may reduce the rate of acid generation, but will not eliminate acid mine drainage. Consequently, bactericides must be used in conjunction with other control techniques for proper environmental protection. Additionally, there is some concern over environmental toxicity of the bactericides which must be applied in strong concentrations to eliminate bacteria.

Bactericides are applied to surfaces of piles and fields using sprayers or hydroseeders. The cost is often several thousands of dollars for each hectare of surface. Because bactericides degrade and are removed by infiltrating water, occasional re-application is necessary. Timed release of bactericide from rubber pellets is reported to extend the lifetime of one application (Kleinmann and Erickson, 1983).

6.8 BASE ADDITIVES

In mining environments with sulphide-rich rock, the potential for acid drainage is based on the relative proportions of acid-producing and acid-consuming materials. Acid-consuming minerals are also known as "alkaline", "basic" or "neutralizing" material. If the potential for acid drainage exists through excess acid-producing material, one potential control technique is the addition of excess neutralizing material, particularly carbonate and hydroxide compounds which produce a neutral to alkaline pH in the associated water (Section 2.3).

The common additives are limestone ($CaCO_3$), lime (CaO or $Ca(OH)_2$), and sodium hydroxide ($NaOH$). These additives are usually used in solid rather than dissolved form because the liquid represents a less concentrated source through solubility constraints. For example, 1 m^3 of $CaCO_3$-saturated water provides approximately 1 kg of $CaCO_3$ whereas 1 m^3 of high-purity limestone provides around 3000 kg of $CaCO_3$. The capability of trace amounts of neutralizing minerals to significantly neutralize acid drainage has been demonstrated by Morin and Cherry (1986) and Dubrovsky (1986).

The success of base additives to control acid drainage depends primarily on (1) the movement of water through the system, (2) the nature of contact of acidic rock or water with neutralizing additives or water, (3) the proportion of excess neutral material, and (4) the type and purity of neutralizing additive.

The movement of water can affect the success of this technique in several ways. For example, the movement of water can affect the rate of acid generation, particularly in a saturated system where the sole source of oxygen is dissolved in and carried by the water. If background groundwater with little oxygen moves upwards into a saturated acid-generating rock pile, the rate of acid generation can be expected to decrease as soon as the oxygen added by disturbance and transport is depleted. Such a situation would lessen the severity of the acid drainage and minimize the necessary quantity of additive for pH control.

A sometimes overlooked complication related to water movement is the consumption of neutralizing additives by pH-neutral water. For example, rainfall passing through a surficial layer of additive and upwelling groundwater passing through a basal layer of additive can usually dissolve some of the additive in excess of the quantity needed to control acid drainage. The rate of water flow will determine the amount of required excess additive required to avoid early depletion of the additive.

The rate and direction of water movement provide the primary connection between acid generation and acid neutralization because these processes cease in the absence of water. This is closely related to the second factor determining successful control, the nature of contact of acidic rock orwater with neutralizing additives or water. The nature of the contact can be divided into three basic scenarios: (1) the additive lies above or "upstream" of the sources of acid generation, (2) the additive is mixed with the acid source, and (3) the additive lies below or "downstream" of the acid source.

The first scenario of acid/neutralizing contact involves the dissolution of additive into ambient water followed by the movement of this water into acid-generating material, such as with rainfall moving downward through a surficial layer of additive inte an acid-generating rock pile. Unless the rate of acid generation is low, this scenario is not effective because sufficient acidity can often be released to the water to overcome the amount of alkalinity of the water contributed by the additive.

The second scenario of acid/neutralizing contact involves the mixing of additive with the acid-generating material. This provides a continual dissolution of additive in response to the acid generation, preventing the development of acid drainage outside the microenvironment around sulphide grains. In fact, laboratory experiments have demonstrated that the rate of acid generation can be slowed significantly, leading to decreased additive consumption, if alkaline water invades the sulphide microenvironment (City Resources, 1988; Helz et al., 1987). As the thoroughness of mixing of additives and acid-generating material decreases, the potential success decreases as the first scenario comes into play and "hot spots" of unhindered acid generation arise. This accounts for the recognized poor success of layers of additive within acid-generating material (City Resources, 1988).

The third scenario of acid/neutralizing contact has the additive downstream of the acid source such as in a basal layer or a collector trench. In this case, the optimum use of additive requires flow of acid water through the additive rather than over the top of the layer, which may not occur in a collector trench lined with a base additive. The elimination of hydraulic short circuits which would allow water to flow around the additive is critical to this scenario. Unlike the second scenario, there is no opportunity for in-situ control of the rate of acid generation.

The third factor determining the potential success of neutralizing additives is the amount of excess additive. The deficit of natural neutralization potential as defined by static and kinetic tests indicates the minimum required quantity of additive as $CaCO_3$ equivalent. This quantity must then be increased to account for the dissolution by ambient precipitation and groundwater (discussed above) and for the encapsulation of additive by precipitates. As acidic water comes into contact with additive and is neutralized, metal compounds such as hydroxides precipitate from the water and may encapsulate the additive, slowing or preventing further neutralization. This is a recognized problem with base additives that has not been overcome at this time, requiring a significantly higher quantity of additive than would otherwise be required. The alternative possibility of forcing alkaline water into the microenvironments around sulphide grains, thereby encapsulating the sulphide minerals, has not be experimentally addressed in detail, but may warrant attention.

The fourth factor determining the potential success of neutralizing additives is the type and purity of additive. The common additives of limestone, lime, and sodium hydroxide differ in their solubilities and, thus, differ in the pH they create upon dissolution. Limestone, a common natural mineral, often raises aqueous pH to around pH 7.0 to 8.5 However, dissolution of limestone into a water is restricted by high calcium concentrations, such as found in gypsum-saturated waters like most acid drainage, through the "common-ion effect". The common-ion effect may limit pH neutralization to pH 5-7. Lime and sodium hydroxide, which are not found in surficial natural environments because of their highly alkaline character, create aqueous pH values approaching 10 and above upon dissolution. Like limestone, their dissolution can also be restricted by the common-ion effect. Nevertheless, lime and sodium hydroxide provide greater neutralization than limestone per unit weight, but the greater unit cost and the environmentally unacceptable alkaline pH detract from the benefit.

The purity of the additive is important for successful control in that the lower the purity, the greater the quantity needed for equivalent neutralization. Lime and sodium hydroxide are manufactured and can be obtained in essentially pure form. On the other hand, limestone is quarried and its purity is often less than 100% as $CaCO_3$. The impurities are other carbonates and other minerals which may not contribute to neutralization. For any choice of additive, a neutralization-potential test identical to the tests carried out on acid-generating material (Chapter 4) should be carried out so that consistency is maintained.

For application of base additives as layers at locations upstream or downstream of the acid-generating material, standard earth-moving equipment is required after the additive is adjusted to a proper grain size to maximize reactivity and geotechnical stability. For mixing of additive and acid-generating material rather than layering, the additive must be brought to the proper grain size to maximize reactivity and ease of application as well as to minimize the potential for additive migration after application. Viable application procedures include slurry spraying, mechanical mixing during disposal, and slurry injection into boreholes following disposal.

The purchase, transport, and application of base additives may cost tens to hundreds of dollars for each m^3. These costs will be relatively low if a minesite is located near a limestone deposit which can be easily quarried and transported.

6.9 COVERS AND SEALS TO CONTROL ACID GENERATION

Covers and seals offer the ability to restrict the access of oxygen and water to reactive wastes. The restriction of water can serve to limit both the formation of acid and the subsequent transportation of the oxidation products into the environment. Only the control of acid generation is considered in this chapter. The exclusion of oxygen is more practical than the exclusion of water for the purpose of acid generation control, as described in Section 6.3.

To limit oxygen and water entry the cover must itself have a low permeability to either air or water and it must not have holes or imperfections through which entry can occur. Should holes or cracks occur in, for example, a cover on a waste rock dump, then oxygen entry takes place as a result of convective flow of air into and out of the dump in response to natural barometric pressure changes and thermal currents through the dump. Cracking of a dump surface, as a result of the large settlements that dumps are prone to over the long term, may result in inflow of surface run-off. The resistance of the cover to cracking, the burrowing effects of roots and animals, erosion and degradation due to weathering and frost action, determines the long term effectiveness of the cover.

A variety of materials may be used to provide surface covers depending on local availability and site conditions. These include different types of soils, synthetic membranes, water and a combination of soil and water resulting in saturated soil or bog conditions, and various other materials such as concrete, asphalt, etc. Alternative cover materials (other than water) and their permeability to water are shown in Table 6.9-1 (Nolan Davis and Associates, 1987).

The most effective means of excluding oxygen is by means of a water cover. The other cover materials are generally more effective as inhibitors of infiltration in the control of ARD migration.

A description of the current technology concerning the use of covers for ARD abatement together with the results of a literature survey are presented by Steffen

Robertson and Kirsten (1988(b)) in a report to the Norwegian State Pollution Control Authority.

6.9.1 Soil Covers

Soil covers show promise as an oxygen inhibitor as can be seen from the published data from the Rum Jungle site in Australia (NTDME, 1986; Bennett et al, 1988; Ritchie and Harries, 1987; Gibson and Partelis, 1988; Verhoeven, 1988). The effectiveness of soil covers as oxygen barriers is influenced by the moisture content maintained in the cover. A cover that can be maintained in a saturated condition will be more effective primarily due to the low diffusivity of oxygen in water and to the absence of dessication cracking. In this regard, composite covers with layers of different soil types to prevent dessication of clay or till have been suggested as potentially beneficial (Magnusson and Rasmuson, 1983;). The reduction in oxygen entry due to soil covers is greatest for very coarse waste rock dumps where oxygen entry by convection is large. Convective transport of air into coarse waste rock is driven by changes in both temperature and barometric pressure and can be large through small holes or cracks in a cover. The long-term performance of soil covers in resisting disruptive forces such as erosion, cracking, frost action, root action and burrowing animals has yet to be proven in the field.

While soil covers inhibit the access of oxygen to the waste and hence may control acid generation, they are generally more effective in controlling infiltration and hence the migration of ARD. To avoid repetition the reader is referred to Chapter 7.0 where soil covers, for the purpose of both oxygen and infiltration control, are discussed in detail.

6.9.2 Synthetic Membrane Covers

Synthetic membranes such as polyvinyl chloride (PVC), high density polyethelene, etc., have the potential to provide covers with an extremely low conductivity to air and hence good oxygen exclusion. These covers may be installed so as to provide good medium-term control, however, long-term degradation, loss of plasticity and ultimate cracking limits the long-term effectiveness of such membranes.

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 affecting 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.

Although the present value of the replacement cost of a synthetic cover on a 100 year basis may be quite small, the monitoring of the condition of the oover, administration of the fund and actual implementation of the replacement are impediments to this approach, despite the fact that it can provide a relatively positive seal to moisture and oxygen penetration.

TABLE 6.9-1

ALTERNATIVE COVER MATERIALS

(After Nolan Davis and Associates, 1987)

Cover Material	Permeability to Water (m/sec.)	Advantages/Disadvantages
Compacted clay	10^{-9} to 10^{-11}	Availability of large quantities problematic in many areas. Subject to erosion, cracking and root penetration. Good sealing if protected and maintained.
Compacted till	10^{-7} to 10^{-9}	As above but generally more permeable.
Compacted topsoil	10^{-5} to 10^{-8}	As above but less robust, more permeable. Questionable longevity.
Peatland bog	10^{-5} to 10^{-6}	Need to maintain in saturated condition. Normally impractical for elevated waste dumps and side slopes.
Concrete	10^{-10} to 10^{-12}	Subject to cracking, frost and mechanical damage.
Asphalt	10^{-20}	As above.
HDPE synthetic	Impermeable	Requires proper bedding and protective cover. Highly impermeable. Lifespan unlikely to exceed 100 years. Subject to root and mechanical penetration.

Other synthetic materials include geopolymers, asphalts and cements such as high volume polypropylene fibre reinforced, sulphate resistant shotcrete. The cost of such synthetic covers are often prohibitively high and these often suffer from cracking and disruption in the long term.

As in the case of soil covers, synthetic membrane covers are more effective in the control of ARD migration and the reader is referred to Chapter 7.0 where these are discussed in more detail.

6.9.3 Water Cover

Water cover is currently the most promising oxygen inhibiting technique and hence the most promising acid generation control measure. The solubility of oxygen in water and the diffusion rate of oxygen through water are both very low. Thus, in the absence of convective transport, the rate of oxygen transport through water is sufficiently slow to be of no concern in terms of acid generation. There is steadily accumulating evidence that underwater disposal of potentially reactive wastes reduces acid generation to negligible levels. (Nolan, Davis & Associates, 1987; Robertson, 1987; McCready, 1987; Senes Consultants Ltd., 1984; City Resources Inc., 1988; Steffen Robertson and Kirsten, 1988(b)). Although oxidation of sulphide and resultant acid generation may not be halted entirely by placing wastes underwater, the rate of acid generation is generally reduced sufficiently to make the impact negligible. Care must be taken when considering placing old wastes, that have previously generated acid, below water because the solution of acid products contained in the waste may occur. The availability of water and the cost of maintaining this cover in the long term are obvious and important site specific criteria that would influence the use of a water cover. Water cover may be achieved by the disposal of waste into natural waters, into man-made impoundments or into flooded underground mine workings and open pits. However, it may not be feasible to achieve water cover on existing deposits and some types of waste facilities.

Despite these limitations, disposal of acid generating wastes under water offers a large number of advantages. Underwater, or subaqueous, disposal is discussed in more detail in Section 6.10.

6.9.4 Saturated Soil or Bog

The effectiveness of a saturated soil layer, for the exclusion of oxygen, has been demonstrated and may be suitable for some categories of waste. Bog conditions can be achieved by the combination of a shallow soil cover with a shallow water cover provided by a water retaining structure. Under these circumstances the waste 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 ARD abatement.

6.10 SUBAQUEOUS DEPOSITION

It has already been identified that the disposal of acid generating waste under water is currently the most promising abatement measure (Robertson, 1987). There is growing field evidence that the disposal of reactive mine wastes under water curtails oxidation to negligible levels (Nolan, Davis & Associates, 1987). This is due to the very low diffusivity of oxygen through water (approximately 2 x 10^{-6} cm^2/sec). This concept can include disposal of waste into natural waters, flooded mine workings, or man-made reservoirs. There are however a number of factors to be considered and possible limitations associated with underwater disposal.

Current legislation (eg. Federal Metal Mining Liquid Effluent Regulations) and the politically controversial nature of lake and marine disposal of mine tailings (acid generating or otherwise) suggests that gaining approval for an application for this method of dispoal for tailings will be difficult to achieve (Knapp, 1989). For this reason it is probably beneficial to fully investigate "on-land" means to achieve water cover first i.e. items 1) and 2) above. Lake and marine disposal of tailings should generally only be considered when all on-land options have been exhausted. This may not necessicarily be the case for other waste types, eg. waste rock.

A study entitled "Subaqueous Disposal of Reactive Mine Wastes" is in the process of being carried out on behalf of the BC AMD Task Force, concurrently with the preparation of this Technical Guide. Information gathered during the preparation of the report on subaqueous disposal was not available and hence is not included in this Draft Technical Guide.

6.10.1 Disposal Into Man-Made Impoundments

Since available evidence indicates that water cover provides the most secure method of acid generation control, consideration should be given to the construction of a water retention facility. The practicality and cost of a man-made reservoir relative to alternative measures is clearly dependent on site specific criteria, for example topography and volume of waste to be stored. The cost of flooding existing waste facilities is likely to be very high. A proposal for the construction of a dam to flood the existing reactive tailings at the Curragh Resources property in the Yukon, Canada, estimated the cost at approximately $50 million (Nolan, Davis & Associates, 1987).

The design of a facility to provide water cover to combined tailings and waste rock may prove beneficial and cost effective for proposed developments. Combined tailings and waste rock disposal may have definite advantages in terms of acid generation control, particularly if the tailings are not acid generating and are discharged at elevated pH (greater than 7). If intimate mixing of tailings and waste rock can be achieved, the permeability of the coarse waste rock would be significantly less than if the rock were placed alone. This has the advantage of reducing potential water movement through the waste rock, (City Resources, 1988).

There are however, limitations and design considerations that may be a disadvantage to man-made water cover facilities. These include:

a) Water retention dams require detailed design of embankment and spillway facilities, careful construction control and maintenance in the long-term. Depending on the site-specific conditions this may prove uneconomical.

b) Reliable water sources must be available to provide a continuous water cover of sufficient depth to avoid exposure of the waste and erosion due to wave action or water flow.

c) Minimum water cover needs to be maintained in low precipitation and drought periods.

d) Water reservoirs may induce unacceptable seepage. If there are other soluble deleterious products in the wastes this may result in increased contaminant loading of the environment. Whether these are significant for the specific project and site conditions has to be determined.

6.10.2 Disposal Into Flooded Mine Workings

Flooding underground mine workings and open pits is a means of controlling acid generation from the exposed rock faces in these facilities. This method also provides a potential disposal area for acid generating waste.

Flooding of worked out coal mines has been successful in the control of acid mine drainage in several instances with acidity reductions of 45 to 99% being reported. The potential benefits to be derived from flooding underground mine workings have been reported for several anthracite coal mines in eastern Pennsylvania (Ladwig et al 1984). Field investigations at the mines, which were allowed to flood some 14 to 20 years ago, revealed that the mine waters which were formerly highly acidic are now slightly alkaline. Sulphate reductions of approximately 54 and 74 percent in mine waters were seen in comparison to 1960's data. In addition, marked decreases in the iron, aluminum, manganese, calcium and magnesium levels were observed.

The disadvantages associated with storage of waste in flooded mine workings are as follows:

a) At single pit operations it is necessary to store reactive waste rock for the life of the mine and to then incur rehandling costs in moving the material back to the pit at a time when the operation is producing no revenue.

b) If all waste removed from the pit is reactive, the swell factor (usually about 30%) will generally result in an excess volume of reactive waste to available underwater storage, particularly as the pit will likely flood to a point less than full.

c) Any sulphides in the pit walls above the final water elevation will oxidize causing a deterioration in water quality unless preventative measures can be applied.

d) Backfilling and flooding precludes future underground development that might be associated with the ore body.

6.10.3 Lake Disposal

Lake disposal is a subaqueous method of disposing of mine waste into an existing natural system. Lakes have been used in the past for the disposal of both acid generating and non acid generating tailings. General case histories of lake disposal of tailings can be identified for the Westmin Mine near Buttle Lake, B.C. (Pedersen, 1983), Surf Inlet Mine near Bear Lake (Errington and Ferguson, 1987), the Mandy Mine near Mandy Lake, Manitoba (Hamilton and Fraser, 1978), a copper mine near Benson Lake, B.C. (Hallam et al., 1974), and the Bluebell Mine near Kootenay Lake, B.C. (Daly et al., 1981). The limited research at these sites suggests no major impact has occurred on lake water quality from the disposal of the tailings due to acid generation. Available evidence indicates that the deposition of acid-generating rock underwater such as into a lake will effectively control the rate of acid-generation. However, lake disposal of tailings also presents other problems related to turbidity and metal mobilization which may affect the biological communities in the lake (Arnesen et al. 1988). Consequently, the environmental concerns related to lake disposal include:

a) toxicity of reagents and heavy metals from the mill process;

b) excessive nutrient additions from the use of explosives;

c) increased turbidity duo to suspended solids causing a reduction in light penetration; and

d) direct physical impact from the placement of waste on the habitat.

Therefore, in selecting a lake for the disposal of tailings or waste rock the relationships between the various site, physical, chemical and biological characteristics should be investigated. As an example, the overall climatic conditions and depth (from bathymetry) will determine the development of density and temperature stratification which in turn may lead to an annual lake turnover. Mixing and stratification control the chemical charactoristics (baseline conditions) and therefore the ability of the lake to assimilate the loading of acidity from the waste material. Finally, all three non-biological characteristics combine to influence the lake ecosystem and its productivity and the tolerance of the ecosystem to disposal of mine materials. Table 6.10-1 outlines some of the characteristics that should be determined to define the suitability of a lake location.

TABLE 6.10.-1

IMPORTANT CHARACTERISTICS IN THE ASSESSMENT OF LAKE AND MARINE DISPOSAL SITES

Site Characteristics	• Proximity of site to mine • Route to lake from mine for tailings transport • Regional climatic conditions • Water/Recreation Use
Physical Characteristics	• Bathymetry • Thermal stratification • Hydrology - turnover and flushing events • Hydrogeology - recharge/discharge characteristics
Chemical Characteristics	• General water quality including pH, buffering capacity, metal concentrations, and alkalinity (seasonal variations) Suspended solids loadings
Biological Characteristics	• Identification of resident fish • Identification of salmonids • Identification of benthos communities • Identification of salmon, crab, and shrimp fisheries (marine) Productivity • Unique Systems Associations

The characteristics of the tailings or waste rock that should be assessed are: acid generation potential, leaching characteristics, and settling properties of tailings. These characteristics will determine the physical and chemical impact of the material on the lake environment. For example, the leaching characteristics will indicate which metals, if any, could dissolve into the overlying lake water.

A more detailed study of underwater disposal sponsored by the B.C. AMD Task Force and MEND Programme is currently in progress. Depending on the findings of the MEND contract on underwater disposal, greater emphasis may be placed on lake disposal in the future. However, the cost of detailed multidisciplinary studies for baseline data and impact predictions on a lake system may limit the economic benefits of this disposal option.

6.10.4 Marine Disposal

Marine disposal is another subaqueous method for disposing of mine waste into an existing natural system, thereby achieving water cover. This option is available for mines situated in close proximity to a marine water body and is popular in some countries such as Norway. Marine disposal in Canada is relatively rare. General case histories of marine disposal in B.C. include Island Copper near Holberg Inlet (Poling, 1979) and Kitsault near Alice Arm (Snow, 1976). Other marine disposal locations include a copper mine in Chile (Castilla and Nealler, 1978) and Norway (Asmund, et al, 1988). As with lake disposal there has been very limited research at the sites related to acid generation control and most of the impacts observed are due to suspended solids and metal leaching from tailings solids.

Many of the site, physical, chemical and biological characteristics of importance in lake disposal (Section 6.10.3.1) must be evaluated at marine disposal sites (Table 6.10.3) however several special factors should be considered. In particular, seawater has a greater buffering capacity and higher alkalinity than fresh water which will affect the chemical interaction between water and tailings. Further, strong tides and currents will disturb the tailings carrying them to other sites and affecting fisheries resources over a wider area.

Some of the waste material's characteristics that should be assessed are: acid-generation potential, settling properties of tailings and metal leaching characteristics. In particular, metal leaching in a saline environment should be considered.

Published research on marine disposal in Canada is lacking and the B.C. ARD Task Force and MEND programme has de-emphasized this option from their study of underwater disposal due to the wider application of other underwater disposal options.

Because many near-shore coastal environments in Canada frequently have fisheries value, reliable evidence for minimal environmental impact of marine disposal would be difficult to obtain. However, there may be local inlets and other coastal areas where fisheries value is minimal and waste disposal may be environmentally acceptable. Otherwise, this option may only be acceptable if long pipelines or barges carry the waste far offshore, but the associated costs would be relatively high.

6.11 CONCLUSIONS

The conclusions that can be drawn from the currently available technology for acid generation control may be summarized as follows:

1. Prevention of the acid generation reactions is the most preferable form of control and should, if at all possible, be the primary long-term approach. The design of ARD prevention at proposed facilities should aim to exclude one or more of the principal ingredients in the acid generation reactions.

2. The exclusion of oxygen from reactive wastes by means of a water cover is currently the most effective acid generation control measure. Water cover (underwater disposal or a saturated soil/bog cover) for preventing acid generation should be evaluated first. Care should be exercised when considering flooding existing waste deposits due to potential high loads of oxidation products within the waste. Proposals to dispose mine wastes, such as tailings, into natural water bodies is often opposed by regulatory agencies and the public for environmental and political reasons. The cost of on-land disposal under water cover, relative to lake or marine disposal, and the environmental implications associated with all methods need to be investigated in full.

3. Control of the acid generation process for abatement of ARD at existing facilities is often not practical, has limited success, or is extremely costly. In these cases, acid generation control techniques may be used to reduce the rate of acid generation in conjunction with control of ARD migration and, if necessary, collection and treatment of ARD.

4. A combination of various measures may produce the most efficient control of ARD for both existing and proposed facilities and in the short or long term. Measures for the control of acid generation should be evaluated in conjunction with control of ARD migration and collection and treatment.

5. Construction methods and extraction processes that result in conditions favourable for preventing acid generation, such as bulk sulphide flotation of tailings, separating high sulphide rock waste, etc. should be considered for

proposed facilities. However, additional control measures are likely to be required.

6. Covers and seals show promise as inhibitors of acid generation provided these are maintained in good order as designed. Certain types of covers and seals are very effective in reducing infiltration of precipitation and soil covers, in particular, are suitable for re-vegetation purposes.

7. The use of bactericides might be a suitable short-term acid generation control measure. It should be remembered that bactericides have a limited life and control only the biological oxidation processes and not chemical oxidation of the sulphides. Additional controls are necessary if the waste has insufficient natural potential to neutralize acid generated by chemical oxidation.

8. Base additives are generally a suitable short-term control measure. In some cases, base additives may be suitable in the long-term, depending on the quantity, type and reactivity of the sulphide minerals. Blending of mine wastes is a form of base addition in areas where limestone or other alkaline strata occur in the overburden. This method has been successfully used in the coal mining industry.

CHAPTER 7: CONTROL OF ARD MIGRATION

7.1 INTRODUCTION

Where acid generation is not prevented, the next level of control is to prevent or reduce the migration of ARD to the environment. Since water is the transport medium for contaminants, the control technology relies on the prevention of water entry to the ARD source. Control of water exit is of little value since in the long term all water entering the ARD source must exit, long term storage being negligible. Water entry may be controlled by:

- Diversion of all surface water flowing towards the ARD source.
- Prevention of groundwater flow into the ARD source.
- Prevention of infiltration of precipitation into the ARD source.
- Controlled placement of acid generating waste.

Diversion facilities usually consist of ditches. Diversion of surface flows, while easily implemented, are often difficult to maintain in the long term. The best long-term solution to such surface flows is to select a disposal site which minimizes the need for diversion. Site selection is generally not an option in the case of open pits and underground operations, while factors other than ARD control may take precedence in site selection for other facilities.

If the ARD source is located over a groundwater discharge area, interception and isolation of the groundwater is very difficult to achieve and maintain in the long term. While measures such as underdrains and sealing layers can be employed, their performance in the long term is questionable. The most effective solution is to select a site which is not located on a groundwater discharge area. Site selection may not be a practical solution.

The secure prevention of infiltration, over the long term, is the most difficult to achieve. Covers of different types may be considered.

Controlled placement of acid generating wastes includes cellular construction of dumps, and tailings deposition methods that increase density and reduce permeability.

7.2 DIVERSION OF SURFACE WATER

7.2.1 General

Surface water can be prevented from entering acid generating facilities by:

- Construction of diversion ditches and berms, and/or
- Site selection to avoid high runoff.

The construction of ditches and berms is generally considered to be a short-term control because of high maintenance costs in the long term. However, maintenance requirements can be reduced for long-term structures by designing for extreme flow conditions with consideration for debris accumulation and providing appropriate erosion protection. While maintenance requirements can be substantially reduced, some form of periodic inspection and maintenance will nevertheless be required. Design flows for diversions can be reduced for facilities such as stockpiles, waste and spoil piles, and tailings deposits by locating these facilities near catchment watersheds. Site selection to reduce runoff may, however, have unfavourable implications to construction and operating costs.

7.2.2 Underground Mines

Surface water inflow is generally not the main source of water in underground mines. Surface water may flow into underground mines through portals, ventilation shafts, or possibly through cracks in the rock that develop by mining induced settlement. At the Balaklala, Mammoth, and Walker mines in California, subsidence has created caved areas that channel surface water into underground workings (University of California, 1988). Diversion can be achieved by ditches and berms in the short term and concrete plugs in the long term.

7.2.3 Open Pits

Diversion of surface water around open pits and strip mines can be achieved with ditches and berms. Creeks and streams may have to be re-routed around these operations. These structures require periodic inspection and maintenance in the long term. The period of inspections depends on the design criteria and level of design adopted. Long-term facilities should be designed for severe flows and possibly with allowance for flow blockage. Erosion protection such as drop structures or rip rap should also be provided. Design flows for short-term structures are generally less severe due to the shorter return period.

Diversion of surface water may also be conducted within the pit limits where one portion of the pit is not an ARD source. This can be achieved with in-pit ditches or berms and by sloping of the pit floor. This is a short-term control and may be expensive to operate because of practical problems and restrictions on mining activities.

7.2.4 Waste Rock Dumps and Spoil Piles

Diversion of surface water around waste rock dumps and spoil piles is achieved with ditches or berms and re-routing of creeks. Requirements for diversion can be reduced by site selection. However, other factors such as haul distance to the dump location influence site selection. Favourable sites are at the crest of slopes or on small plateaus and near to the upstream end of a watershed, however, there may be unfavourable cost implications with these locations. As in Section 7.2.3 long-term facilities will require more stringent design than short-term facilities and will require some form of maintenance.

Diversion of surface water around acid generating dumps cannot be avoided by the use of under-flow through base drains. Even if the drain is composed of non-acid generating waste the water flowing through the drain will become contaminated by water which has infiltrated through the overlying waste. Separation of these waters with barriers in the dump is not considered practical. Differential settlement within the dump would damage or destroy such a barrier.

7.2.5 Tailings Deposits

The approach and methods for diversion of surface water around tailings deposits are the same as those described for waste rock dumps, Section 7.2.4. However, natural topography is important in site selection for tailings dams. The most suitable location for a tailings dam, in terms of minimizing construction materials, is most often one that requires substantial diversion of surface water. Construction cost and the cost of long-term maintenance of diversion facilities needs to be optimized in site selection. Other forms of ARD control also need to be considered. For example, the control of acid generation by means of water cover eliminates the need for surfacewater diversion. In the case of acid heap-leach operations, the inherent low pH of the material and pore water is an important consideration in the design of control measures.

7.2.6 Stockpiles and Spent Heap-leach Piles

The methods for diversion of surface water around stockpiles and spent heap-leach piles are the same as those described for waste rock dumps, Section 7.2.4. Proximity to the mill site will usually be a principle factor in site selection for stockpiles. The presence of chemicals from the leachate used for mineral extraction in heap-leach operations may require a more stringent design than required for stockpiles.

7.3 GROUNDWATER INTERCEPTION

7.3.1 General

When ground water enters, or comes into contact with acid generating waste this provides a transport medium for the contaminants. The entry of ground water into waste facilities may be prevented by:

- Interception or isolation of ground water before it enters the waste, or site selection to avoid groundwater discharge into the waste.

The objective is to minimize ground water contact with acid generating material or other water which has become contaminated by acid generation.

All collection and interception methods are prone to failure in the long term. Therefore, selection of a site which avoids groundwater discharge is the best method of control. Site selection is not an option for underground mines and open pits but should be considered for waste dumps, spoil piles, tailings deposits, stockpiles and spent ore piles. Any potential storage or waste disposal site will be located in either a groundwater recharge area, a groundwater discharge area, or an area with sufficient seasonal variation in groundwater level to be both a recharge and discharge area, depending on the time of year. This last case may be most severe in terms of environmental impact. The performance and cost of different groundwater interception and isolation methods vary over a wide range, depending on hydrogeologic and other site specific parameters.

7.3.2 Underground Mines

Groundwater interception or control can be achieved by various methods for underground mines, depending on the site geohydrology. Regional dewatering around underground openings can be achieved with wells from surface or perimeter drainage galleries. Localized control of ground water in conductive aquifers encountered by mining can be achieved by grouting the aquifer or dewatering with drain holes. In any method where the water is collected within the mine, efforts will be required to prevent it from contacting acid generating material. Exploration drill holes can be a large source of groundwater flows into underground workings. Inflow through boreholes can be controlled by properly grouting and sealing the holes after drilling.

7.3.3 Open Pits

Dewatering around open pits is a common practice for improving slope stability. Groundwater interception and removal can be achieved with perimeter wells, drainage adits and horizontal drainholes. However, an important consideration is that ARD could develop in wells and adits if these pass through acid generating rock and oxygen is present. A means of preventing this acid generation is to keep the adit, well, or drain flooded hence minimizing oxygen access.

Horizontal drainholes are not well suited to groundwater interception for the prevention of ARD if these drainholes flow into the pit where contact with acid generating rock is likely to occur. A pipe collection system can be installed to prevent ARD contamination. However, a low level of performance should be anticipated because of damage caused by mining activities, and bench and pit slope failures. Groundwater isolation by grouting may be effective for areas of high inflow.

All of these methods are short-term controls although drainage adits which drain away from the pit could provide long-term isolation.

7.3.4 Waste Rock Dumps and Spoil Piles

Groundwater interception or isolation under waste rock dumps and spoil piles will generally be difficult to implement and maintain. Site selection which avoids groundwater discharge is the best method of preventing ground water flowing into dumps. If a waste facility is located in a groundwater recharge area, and acid generation and migration are not controlled, there is a risk of groundwater contamination occurring. In some cases it may be necessary to situate the dump on a groundwater discharge area and provide measures for acid generation control or facilities for collection and treatment.

7.3.5 Tailings Deposits

Groundwater interception or isolation at tailings impoundments is generally not practical. Avoiding groundwater discharge by site selection is difficult because the best disposal sites are usually valleys, which are often also groundwater discharge areas.

7.3.6 Stockpiles and Spent Heap-leach Piles

Groundwater isolation under stockpiles and heap-leach piles can be achieved with engineered barriers. These are not likely to provide long-term control. Isolation can be achieved by placing the stockpile on an impermeable layer, overlying a drainage layer to remove groundwater inflow. Integrity of the impermeable layer is critical for successful isolation. If damaged, the impermeable layer is very difficult to repair without removing the stockpile.

7.4 COVERS AND SEALS TO CONTROL INFILTRATION

7.4.1 General

The transport medium for contaminants is water and the principle source of this water is infiltration of precipitation. The control of infiltration is therefore important in controlling ARD migration. The most practical way of controlling infiltration of precipitation is by means of low permeability covers or seals. Soil and synthetic materials are commonly used to construct covers. These can be applied to rock faces in open pits or underground mines, waste rock dumps, spoil piles, tailings deposits, stockpiles and heap-leach piles. The length of time during which control is required is an important consideration in selecting the most appropriate cover material or combination of materials.

7.4.2 Soil Covers

Soil covers are introduced in Chapter 6.0 of the guide (Section 6.9.1) in the discussion of methods to inhibit oxygen access to wastes. It is concluded that, while soil covers do show promise as oxygen inhibitors, they are generally more effective in controlling infiltration of precipitation. The effectiveness of soil covers as inhibitors of infiltration depends on factors such as climate, cover design and construction. The mechanisms of infiltration, design of covers, and effectivaness of soil covers as barriers to infiltration are discussed in this section of the guide.

7.4.2.1 MECHANISMS OF INFILTRATION

7.4.2.1.1 Water Flow in Unsaturated Soils

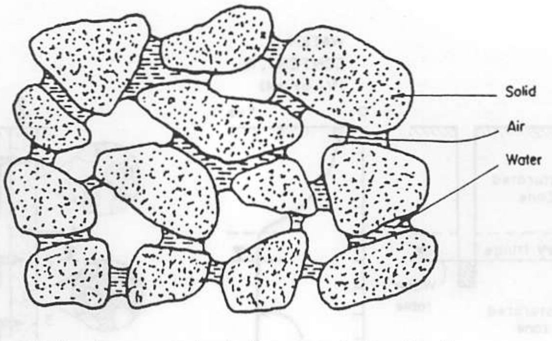
Water transport in the cover material and in the waste underneath takes place under generally unsaturated conditions (Collin, 1987). This implies that the porous material is partly filled with air. Flow under unsaturated conditions is considerably smaller than under saturated conditions. Water statics and dynamics in the unsaturated zone are also of utmost importance for the diffusional transport of oxygen from the surrounding air since a high moisture content in the cover material is needed to restrict this transport. A general description of water flow in unsaturated materials follows (substantially summarized from Siwik et al, 1987). For a more detailed account see Collin (1987), Steffen, Robertson and Kirsten (1986a), Rasmuson (1978), or Bear (1972 and 1979).

One of the fundamental relationships that describes fluid flow through porous media is Darcy's Law:

$$v = -k \frac{dh}{dx}$$

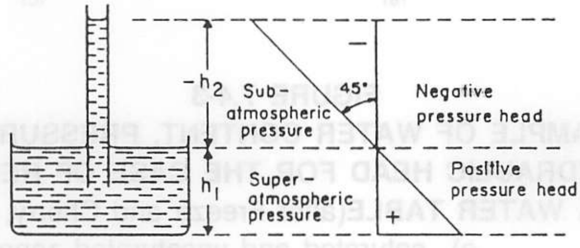
Darcy's Law is a linear relationship between flow velocity (v) and the driving force, the hydraulic gradient (dh/dx), with the proportionality coefficient being the hydraulic conductivity (k). This relationship holds true for both saturated and unsaturated conditions.

Whereas the water in a saturated soil is under positive pressures relative to atmospheric (gauge pressure), the water in an unsaturated soil is held in the pores under negative pressures (tension) caused by the surface tension between water and air (see Figure 7.4-1). This soil-water pressure head varies with water content (see Figure 7.4-2) because smaller pores hold water under greater tension than larger pores, due to the greater curvature of the water menisci. Figure 7.4-3 shows the result of this phenomena relative to the water table. In the saturated soil, the hydraulic conductivity (k) is a constant; however, in the unsaturated zone, k varies with pore water pressure head, (p_w) (see Figure 7.4-4) and thus with the water content (w). As the pore water pressure head becomes more negative (drier soil), more pores drain and no longer contribute to the flow of water, thereby reducing the hydraulic conductivity of the soil.



Water in an unsaturated coarse-textured soil.

(a)



(b)

FIGURES 7.4-1
WATER IS HELD IN UNSATURATED POROUS MEDIA UNDER NEGATIVE PRESSURE HEAD (after Hillel, 1980)

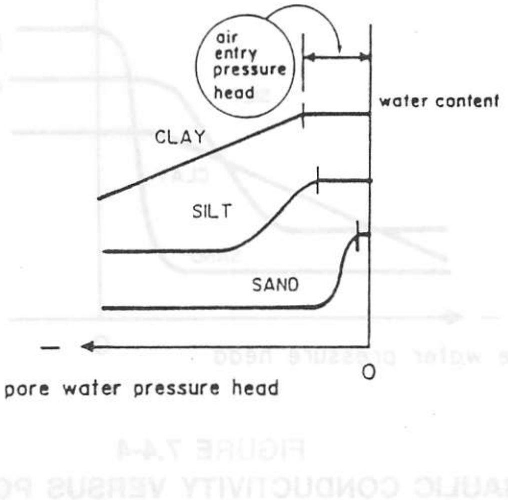


FIGURE 7.4-2
SOIL-WATER CONTENT VERSUS PRESSURE HEAD (IGNORING HYSTERESIS) (after Freeze and Cherry, 1979)

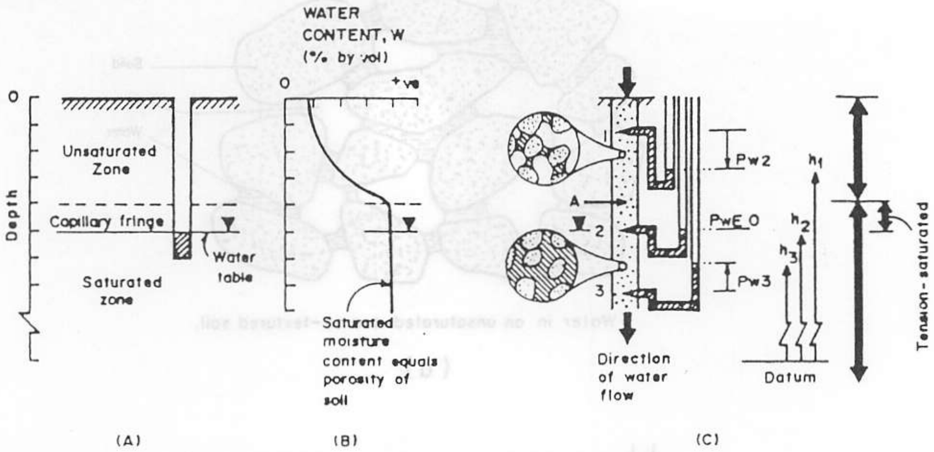


FIGURE 7.4-3
AN EXAMPLE OF WATER CONTENT, PRESSURE HEAD AND HYDRAULIC HEAD FOR THE CASE OF RECHARGE TO A WATER TABLE(after Freeze and Cherry, 1979)

a) saturated and unsaturated zones
b) profile of moisture content versus depth
c) pressure head and hydraulic-head relationships; insets; water retention under pressure heads less than (top) and greater than (bottom) atmosphere

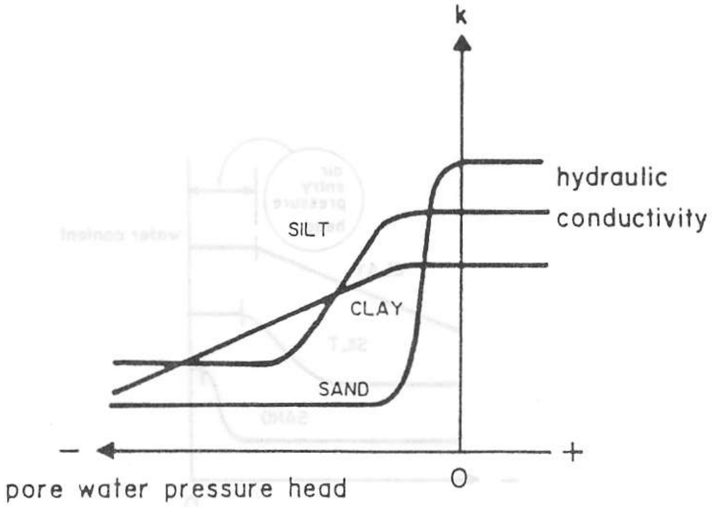


FIGURE 7.4-4
HYDRAULIC CONDUCTIVITY VERSUS PORE-WATER PRESSURE HEAD (after Freeze and Cherry, 1979)

a) uniform sand
b) silty sand
c) silty clay

A clearer conceptual appreciation of the difference between water flow in saturated and unsaturated soil can be achieved by inspecting Figure 7.4-4. This diagram shows the variation of the hydraulic conductivity (k), the pore water pressure head (or simply noted as pressure head (p_w) and the water content (w), along a horizontal soil core conducting water under steady-state conditions.

The non-linear characteristics of the unsaturated flow system result in it being more difficult to accurately model water flux.

$$\frac{\partial w}{\partial t} = \frac{\partial}{\partial x} \left[k(p_w)\frac{\partial p_w}{\partial x} \right]$$

where:

w = the volumetric water content
p_w = the pressure head
k = the unsaturated hydraulic conductivity.

Traditionally, the subject of unsaturated flow of water in soil has been dealt with by subdivision into narrower topics with infiltration, drainage and evapotranspiration being prominent. The following is a brief discussion of these three topics for uniform soil profiles (such as a single layer engineered soil cover). The textbooks by Hillel (1980a,b) provide a more in-depth discussion than will be presented here and are good sources of further references on these topics.

The rate at which water infiltrates into an unsaturated soil when water is applied to the soil surface at atmospheric pressure (infinitesimal ponding depth) is termed the infiltrability. If water is ponded to some depth on the surface of the soil, the infiltration rate will exceed the infiltrability. The infiltration rate will be less than the infiltrability if the water is applied at a rate lower than the infiltrability. Even when water is applied at atmospheric pressure, the infiltrability is not constant. As shown in Figure 7.4-5 the infiltrability decreases with time until, at long time, a steady infiltrability value is approached. A further complication arises in that the infiltrability depends on the initial water content of the soil as shown in Figure 7.4-6. For the purpose of introducing concepts of infiltration into a uniform soil layer, we will restrict the remainder of the section to infiltration of shallow ponded water on the soil surface. Figure 7.4-7 shows plots of hydraulic head and degree of saturation versus depth for continuous shallow ponding of water on the soil surface.

As the wetting front reaches a given depth in the soil, the hydraulic head increases with time, as does the degree of saturation. When time is relatively large, the hydraulic gradient approaches the unit gradient with the pressure head equal to zero throughout the profile and the change in hydraulic head is due to the change in elevation head only. At that time, the water flux density (specific discharge) is numerically equal to

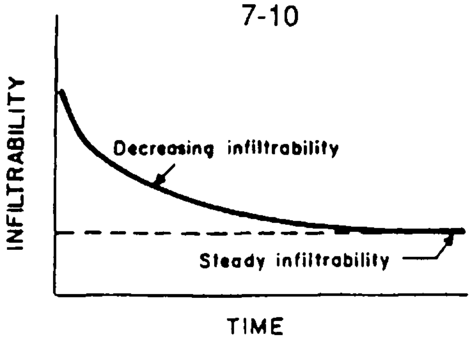


FIGURE 7.4-5
INFILTRABILITY VERSUS TIME (after Hillel, 1980b)

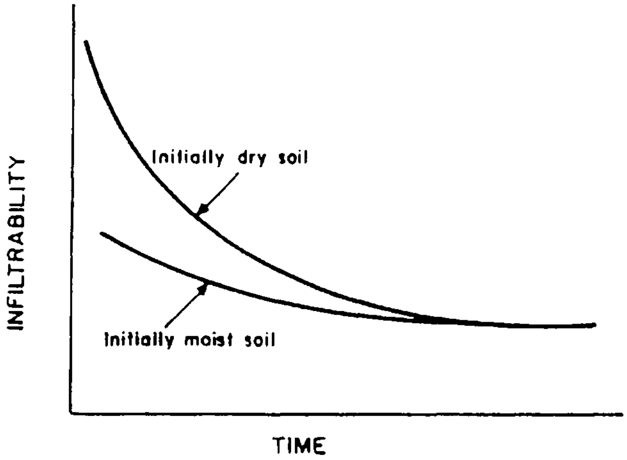


FIGURE 7.4-6
INFILTRABILITY VERSUS TIME FOR AN INITIALLY DRY AND INITIALLY WET SOIL (after Hillel, 1980b)

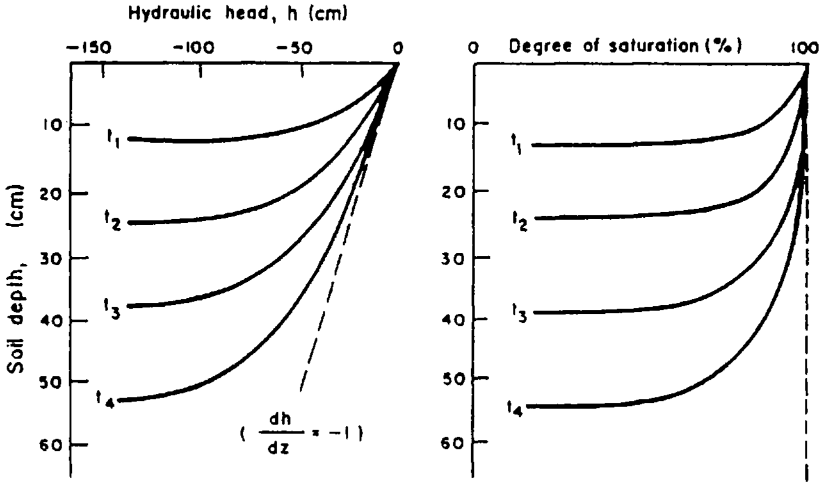


FIGURE 7.4-7
WATER CONTENT PROFILES (AT RIGHT) AND HYDRAULIC HEAD PROFILES (AT LEFT) AT DIFFERENT TIMES DURING INFILTRATION (after Hillel, 1980b)

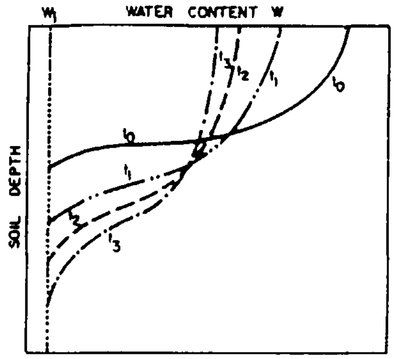


FIGURE 7.4-8
THE CHANGING MOISTURE PROFILE IN A MEDIUM-TEXTURED SOIL DURING REDISTRIBUTION FOLLOWING AN IRRIGATION.
The moisture profiles shown are for T=O, 1, 4 and 14 days after Irrigation. Wi Is pre-irrigation (antecedent) water content (after Hillel, 1980b)

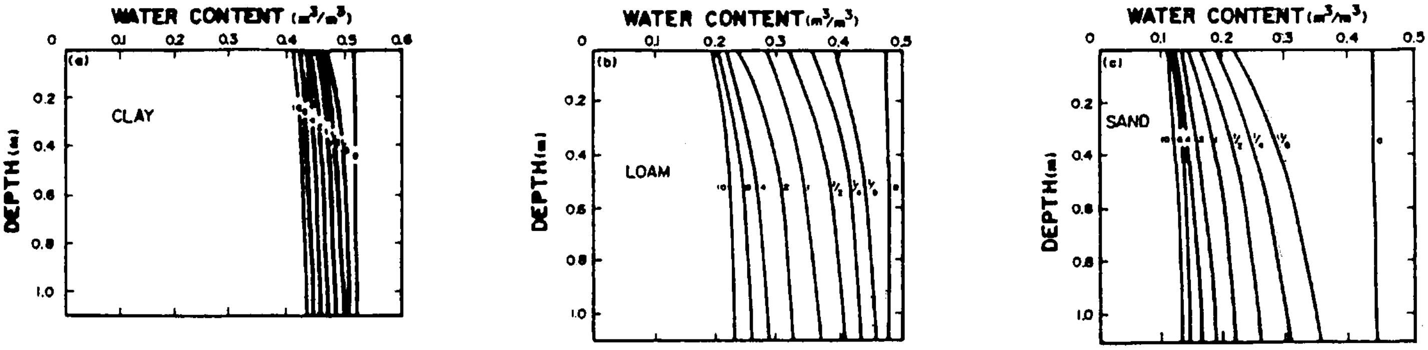


FIGURE 7.4-9
THE CHANGING SOIL MOISTURE DISTRIBUTION DURING DRAINAGE FROM INITIALLY-SATURATED UNIFORM PROFILES OF (A) CLAY, (B) LOAM, AND (C) SAND. THE NUMBERS INDICATE DURATION OF THE PROCESS (DAYS) (after Hillel, 1980b)

the hydraulic conductivity. Under these conditions water will not infiltrate at a rate greater than the hydraulic conductivity unless significant ponding occurs on the surface.

When water is applied to the soil surface for a relatively short period of time, the water redistributes in the soil profile as shown in Figure 7.4-8. The actual water content at a given depth and time is dependent on the soil water characteristic curve of the particular soil in question. This type of infiltration and redistribution would occur in a waste cover after a rainfall event.

The changing soil moisture distribution during drainage of a uniform soil profile that is initially saturated is shown in Figure 7.4-9. Comparing the sand, loam and clay soils, we see that fine grained soils maintain higher water contents (therefore lower air filled porosity) than coarse grained soils drained for the same period of time. This is due to both the lower conductivity and smaller pores in the clay soil.

This relationship between soil textures is also apparent in plots of volumetric water content versus time at a depth of 41 cm (arbitrary) shown in Figure 7.4-10. Figure 7.4-10 also shows the additional effeot of simultaneous evaporation from the soil surface. Covers in western Canada would be saturated in the spring during and immediately after snow-melt, and would likely exhibit the type of drainage shown in Figures 7.4-9 and 7.4-10.

The water content versus soil depth profile caused by evaporation at a bare soil surface is shown in Figure 7.4-11. If vegetation was present, the plant roots would be extracting water form the soil to be transpired to the atmosphere. The combined effect of evaporation and transpiration is termed evapotranspiration. The presence of vegetation results in a greater loss of water form the soil and drying of the soil to a greater depth, dependent on plant type and time of year. It also tends to disrupt drainage and this increases ponding and infiltration. There are a number of other factors which influence the infiltrate rate, such as the texture and compactness of the soil surface, texture and irregularity and slope and the resulting ponding, duration and intensity of rainfall. Prediction of infiltration rates through uniform soils is therefore extremely complex. Cracks, root holes, burrows and soil variability makes this even more complex.

7.4.2.1.2 Flow Through Layered Soils

By placing layers of coarse and fine grained soils above each other it is possible to develop infiltration barriers and drainage layers. These complex engineered covers are reviewed in greater detail in Section 7.4.3 of the Guide.

A concept for a complex layered soil oover for sulphidic mine waste deposits is described by Magnusson and Rasmuson (1983). The waste is covered with, from below, a coarse-grained layer, a fine-grained layer, another coarse-grained layer and a till layer (Figure 7.4-12). This cover system includes capillary barriers which are formed at the interfaces between coarse-textured and fine-textured soils. The capillary barrier concept involves the principles of unsaturated flow between materials of different textures. If a fine-textured soil overlies a coarse-textured soil and both materials are initially dry, water infiltrating from the surface will not move into the coarse-textured soil until the capillary potential in the fine-textured soil approaches zero (i.e., until the fine-textured soil is at, or near, saturation). The capillary barrier can also be placed above the fine-grained barrier layer thereby preventing capillary water transport upwards from the fine-textured soil.

Drying of the fine-grained layer, causad by capillary water transport upwards and downwards from the fine-grained layer, is prevented. The lower coarse-grained layer may also form a capillary barrier at the top of the waste sand, thereby preventing capillary transport of contaminated water from the waste sand to the soil cover. The upper coarse-grained layer may form a capillary barrier at the bottom of the till layer. Furthermore, the upper coarse-grained layer may act as a drainage layer resulting in an increased lateral water drainage and hence a decrease of infiltration.

A conceptual moisture content profile in such a cover system is shown in Figure 7.4-13. This design of the cover gives very good possibilities for the reductien of both the oxygen transport and the water percolation through the cover. For economic reasons, however, a simpler soil cover would be preferred.

7.4.2.1.3 Modelling Infiltration

Modelling of water flow in unsaturated soils is complicated by the non-linear nature of the interrelationships of the soil water properties, and by the fact that steady-state flow conditions are seldom achieved in the field.

Alternative methods for the modelling of infiltration have been reviewed by Steffen, Robertson and Kirsten (1986a).

The Hydrological Evaluation of Landfill Performance (HELP) model developed by the U.S. Army Corps of Engineers is an extremely useful tool for a first evaluation of the relative benefits of alternative cover layers with and without vegetation cover. The large potential differences in infiltration between the different cover types is apparent. The HELP model assumes saturated soil conditions at the commencement of precipitation. Clearly this is an approximation leading to considerable inaccuracies under certain circumstances.

The TRUST model was developed at Lawrence Berkeley Laboratories and is well described by Reisenauer et al, (1982). This suite of programs has been extended to allow more effective and convenient applications to cover design by McKeon et al, (1983). It was used by Collin, (1987), for the evaluation of the effectiveness of the complex cover shown in Figure 7.4-12. While the program allows for partially saturated flow the determination of input variables to adequately allow for the seasonal and extreme variations found in practice still renders the accuracy of the answers questionable.

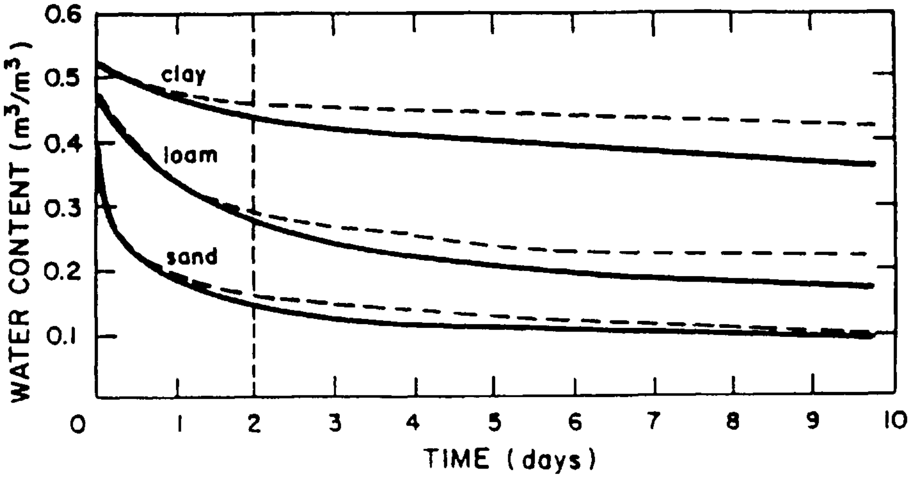


FIGURE 7.4-10
WATER CONTENT (W) AT DEPTH OF 41 cm AS FUNCTION OF TIME IN INITIALLY-SATURATED UNIFORM PROFILES OF SAND, LOAM, AND CLAY. DASHED LINES-DRAINAGE WITHOUT EVAPORATION; SOLID LINES-SIMULTANEOUS DRAINAGE AND EVAPORATION (after Hillel, 1980b)

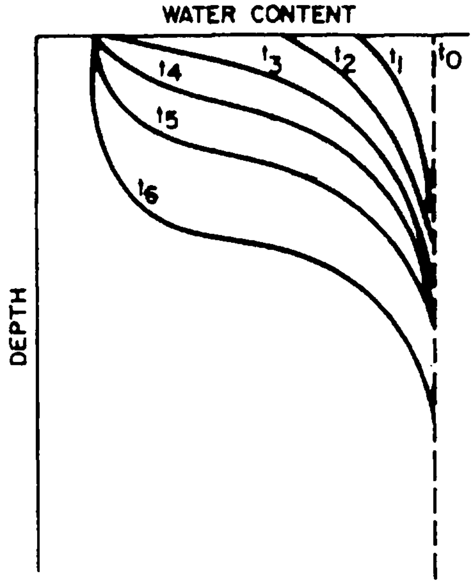


FIGURE 7.4-11
WATER CONTENT VERSUS DEPTH SHOWN FOR VARIOUS TIMES DURING EVAPORATION FROM AN INITIALLY SATURATED SOIL

A model that simulates unsaturated conditions, called SOILMOIST, has been developed for the US Uranium Mill Tailings Remedial Action Program and this may yield more realistic results under certain circumstances.

A limitation that all the models suffer from is the inability to anticipate and include the effects of layer disruptions such as:

- settlement causing drainage disruption and ponding
- cracking due to settlement or desiccation
- root holes
- burrowing channels formed by insects, animals and man
- frost action effecting permeability and drainage
- erosion
- disruption of drainage by vegetation
- clogging of drains due to frost or root action

These disruptions may be more severe with complex layered covers than with simple homogeneous covers.

7.4.2.2 SIMPLE COVERS

In the interest of minimizing cost, a simple, single layer, soil cover is preferred. A fine textured soil, such as clay or silt, is required to limit infiltration. To effectively limit oxygen transport it is necessary to maintain the layer at a high moisture content. A single soil layer, however, is limited in its effectiveness for the following reasons.

- Without capillary barriers, a simple soil cover is prone to large seasonal variations in moisture content. This could result in desiccation cracking and hence an increase in permeability. In addition, decreasing the moisture content of the soil increases the rate of oxygen diffusion. These seasonal variations are greatest near the surface and their effect is therefore greatest for thin covers. For single layer soil covers to be effective they need to be relatively thick to maintain a saturated zone during the dry season. The cover thickness required is probably a function of the climate but is likely to be of the order of 2 m for regions such as B.C.

- The fine-grained soils required to limit infiltration may be frost susceptible. Ice segregation may result in degradation of the cover and increased permeability. Frost heave may also make the surface of the cover irregular, allowing ponding and increased infiltration.

- A simple soil cover does not have the ability to prevent moisture being sucked up from underlying tailings by capillary action. Likewise, it does not limit the migration of salts from the tailings to the surface due to surface evaporation and transpiration.

- A simple, single layer fine-grained soil cover may not be able to adequately withstand wind and water erosion or burrowing and root action. Some form of erosion protection, such as vegetation or rip-rap is normally required.

These limitations on the effectiveness of a single soil layer can be overcome by using complex covers, as described in the next section.

7.4.2.3 COMPLEX COVERS

The effectiveness of a soil cover is greatly improved by adopting a complex cover design consisting of several layers, each performing specific functions to improve water and oxygen exclusion and long-term stability. These layers and their specific functions are described below. A typical complex cover design is illustrated in Figure 7.4-12 (Rasmuson and Eriksson, 1987).

1) Erosion Control Layer

Erosion protection can be provided by vegetation or by a layer of coarse gravel or rip-rap. The establishment of vegetation on the waste dumps is desirable for aesthetic and land use reasons. Therefore, revegetation is usually the most desirable method of providing erosion control. However, where revegetation is not practical or will not sufficiently control erosion, coarse gravel or rip-rap may be required.

Studies for uranium tailings deposits in Canada (Steffen, Robertson and Kirsten, 1986a) indicated that forest cover would adequately control sheet and rill erosion, and wind erosion, but no methods of analysis are available to assess the effectiveness of vegetation on gully erosion. A vegetation cover design guide has been prepared for U.S. uranium tailings deposits by Beedlow (1984). While this design guide illustrates and discusses many of the aspects relevant to vegetation cover design, it is not necessarily directly applicable to B.C. mine waste deposits. A similar guide for the design of rip-rap erosion protection has been prepared by Walters (1982).

A special study on vegetative covers was recently carried out as part of the Uranium Mill Tailings Remedial Action Project (UMTRAP) in the United States (U.S. Department of Energy, 1988). This study investigated the use of vegetation to stabilize uranium tailings, and specifically includes the use of vegetation to intercept infiltration. The principal finding of the study is that properly developed plant communities on complex soil covers can be effective in stabilizing covers and controlling infiltration on topslopes of waste piles. The study showed that the appropriate vegetative cover will adapt to climatic change, will repair itself after severe disturbances such as fires and droughts and will persist indefinitely with little or no maintenance. The plants were found to protect topslopes against sheetwash erosion, however, resistance to gully erosion depends more on the overall pile configuration rather than on the vegetation and soil.

layer is to provide a sufficient barrier to enable the overlying coarse-grained layer to drain infiltration and prevent ponding of water.

5) Lower Capillary Barrier

Rasmusson and Eriksson (1987) investigated the use of capillary barriers, beneath the infiltration barrier, to reduce infiltration. The principle is that, if negative pore-water pressure is maintained in the low permeability material at the interface with the underlying coarse grained capillary barrier, infiltration into the lower layer would be prevented. They found that this would only be effective if ponding on the low-permeability layer does not occur, which would be difficult to achieve in practice. However, for soil covers over fine grained waste deposits such as tailings, a capillary barrier beneath the infiltration barrier may be useful in preventing suction of contaminated pore water from tailings up into the cover in dry periods.

The long-term performance of a complex soil cover could be greatly reduced if fine-grained materials are allowed to migrate into the coarse-grained layers. Filter layers should be considered.

6) Basic Layer

A basic layer could be incorporated into the design to reduce the pH of infiltrating water and therefore acid generation rates. Alkaline materials such as limestone could be spread over the surface of the waste before placing the cover or mixed into the cover layers.

Limestone is commonly mixed with waste rock during placement at coal mines with great success and research is being done on the addition of phosphate rock (Chiado et al, 1988). Base addition to control acid generation is discussed in more detail in Chapter 6.0, Section 6.8. 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 may be insufficient to control ARD. 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 addition of a basic layer would not significantly reduce acid mine drainage where unsaturated conditions predominate, such as in waste piles. It would be more beneficial in saturated tailings, and might be usefully employed in tailings impoundment covers.

7.4.2.4 The Effectiveness of Soil Covers

Information on the relative effectiveness of soil covers in controlling acid mine drainage may be obtained from the results of mathematical model simulations of covers, and from the results of monitoring a limited number of actual covers.

Acid flux reductions calculated by Steffen Robertson and Kirsten (1987) using the RATAP model (SENES and Beak, 1986) for some simple cover types are tabulated in Table 7.4-1, and illustrate the theoretical effects of rip-rap, soil and limestone covers.

TABLE 7.4-1

EFFECT OF COVER OPTIONS ON ACID GENERATION

(After Steffen Robertson and Kirsten, 1987)

Tailings Cover Type	Cover Thickness (m)	Sulphate Flux (moles/m^2/year)	Reduction of Acid Flux (%)
None	-	76.65	-
Waste Rock (inert)	0.6	59.37	22.5
Depyritized Tailings (soil)	2.0	21.06	72.5
Depyritized Tailings (plus) Limestone (soil)	2.0	7.9	89.7

The first cover type considered in Table 7.4-1 is to provide a non-acid generating waste rock (rip-rap) cover. The primary consequence of this option is a reduced annual summer temperature in the tailings. The results from the mathematical model show a corresponding reduction in acid flux. The effect of a soil cover, in the form of inert tailings, is to lower the temperature as well as raising the water table and hence reducing acid generation. The third type of cover demonstrates the use of limestone to control the pH and hence reduce acid generation. It is important to note that the above example is based on the results obtained from a specific mathematical model and are therefore subject to the assumptions, input and method of calculation adopted for that model. Nevertheless, the example serves as an illustration of how the relative effectiveness of different covers may be evaluated. Figure 7.4-14, developed from results using the RATAP model, illustrates the effect of depth of a till cover on typical tailings from the Elliot Lake project for various depths of unsaturated tailings (Steffen Robertson and Kirsten, 1987).

Figure 7.4-15 shows the effects of various types of covers on infiltration rates as predicted by the HELP model for acid generating tailings at Elliot Lake (Steffen Robertson and Kirsten, 1987). The results show considerable benefit of a complex cover design over simple covers, and benefit of adding a synthetic liner.

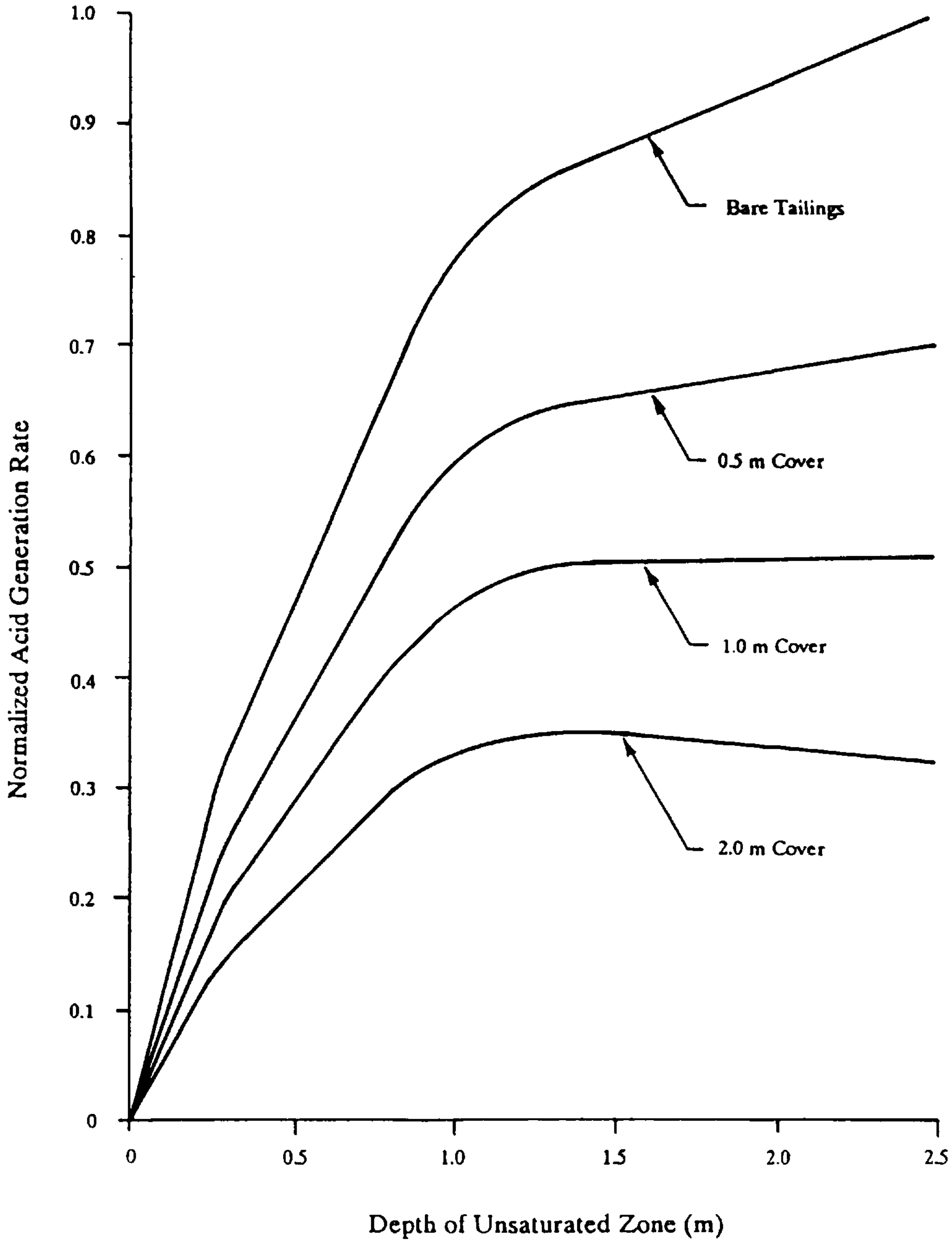


FIGURE 7.4-14
EFFECT OF TILL COVER DEPTH
ON ACID GENERATION RATE IN TAILINGS
(After Steffen Robertson & Kirsten, 1987)

The results of the infiltration modelling runs are discussed below.

i) Bare Tailings (Figure 7.4-15 (A))

Bare tailings can be expected to have high runoff rates, modest evapotranspiration losses and substantial net infiltration or seepage. With an unvegetated surface, the run-off can be expected to be quite high. For the example model runs, runoff, evaporation and seepage rates account for 19%, 63.5% and 17.5% of the annual precipitation respectively. When the tailings permeability was increased by a factor of 3, runoff rates decline by 3% and seepage rates increase by 3%.

ii) Vegetated Tailings (Figure 7.4-15 (B) and (C))

Vegetation has a marked effect on the water balance at a tailings site. With the growth of a "moderate" vegetation cover, runoff rates decrease from 19% to 9% of the annual precipitation while evapotranspiration increases from 63.5% to 73.6%. The major finding is that seepage rates are not changed. With a good vegetative cover, runoff rates are again reduced further to 3.3% of the annual precipitation. Although evapotranspiration rates are increased, this may not offset the reduced runoff. This is not to conclude that this phenomenon is universally applicable to all sites.

iii) Soil Cover (Figure 7.4-15 (D) and (C))

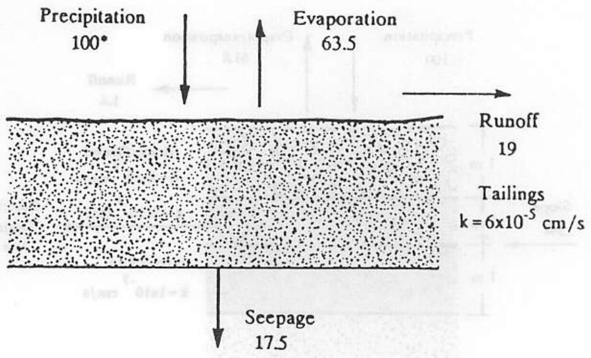
Direct application of soil to the tailings area surface may have mixed effects. If the soil retains its low permeability, runoff will increase substantially and seepage rates will be greatly reduced. A compacted till cover with a permeability of 2 x 10^{-8} m/s will reduce seepage rates to less than 2% of the annual rainfall. If this cover cracks and weathers (as is expected) infiltration rates increase substantially. The example indicates that if the effective permeability of the cover increases to 1 x 10^{-4} m/s, seepage rates exceed those for bare tailings. The increased permeability results in a major reduction in the rate of surface runoff.

iv) Rock/Gravel Cover (Figure 7.4-15 (F))

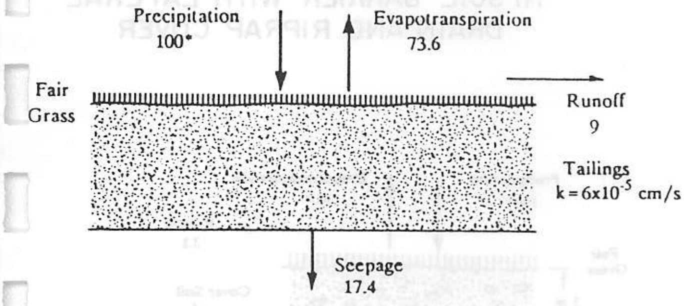
Rock or gravel is often applied to stabilize the surface of a tailings area. This pervious layer effectively eliminates runoff and therefore can substantially increase infiltration rates. For the modelled case, the rock/gravel cover increases seepage rates from 17.5% to 37.8%. This is more than a factor of 2 and further demonstrates how the permeability of the surface layer can affect the overall amount of seepage produced.

v) Engineered Cap (Figure 7.4-15 (G), (H) and (I))

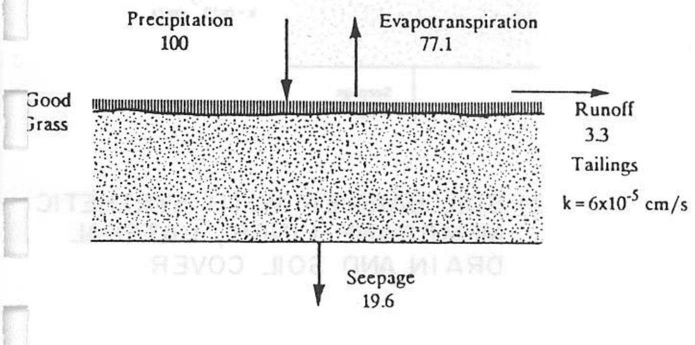
A properly constructed engineered cap can greatly reduce infiltration rates. The example modelled includes a cap with one metre of soil for frost protection and vegetation, 0.3 metres of lateral drainage layer, and one metre of a low permeability



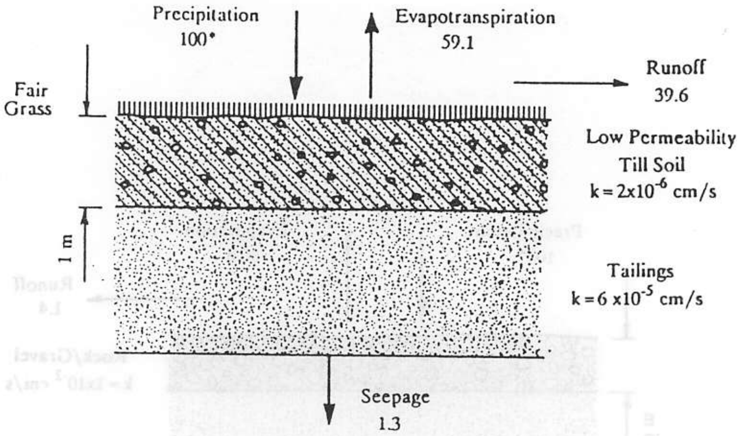
A) NO COVER



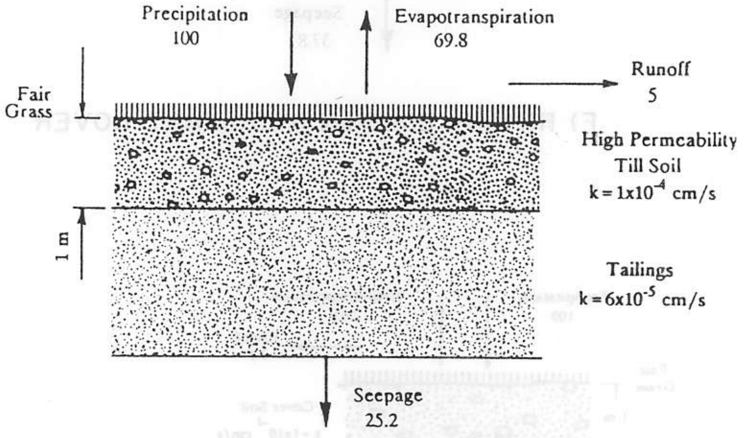
B) MODERATE VEGETATION COVER



C) GOOD VEGETATION COVER



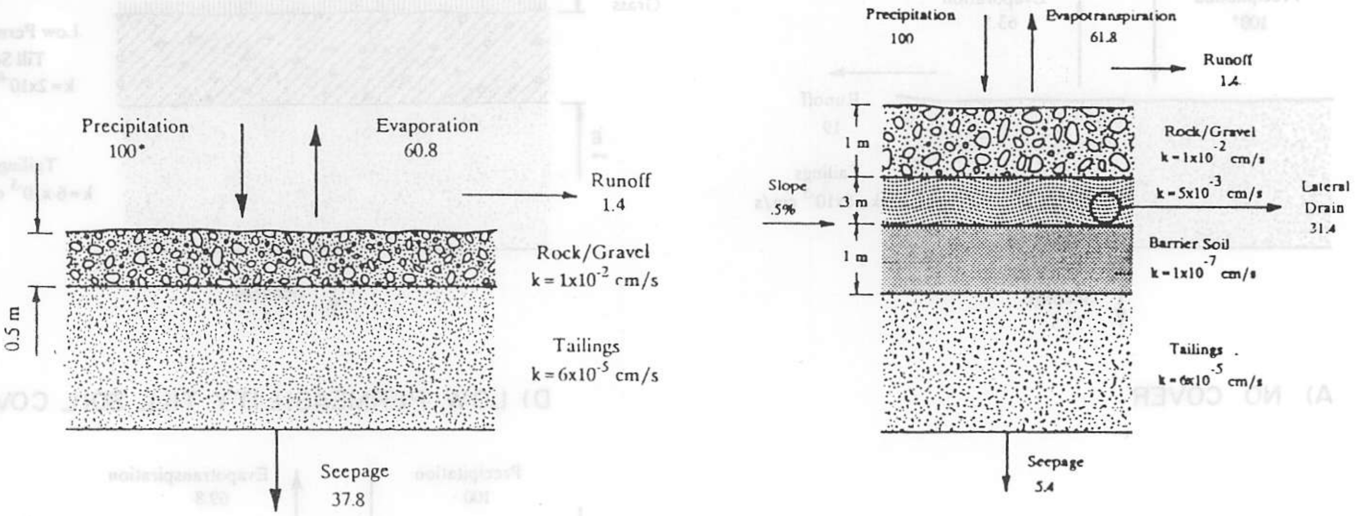
D) LOW PERMEABILITY TILL SOIL COVER



E) HIGH PERMEABILITY TILL SOIL COVER

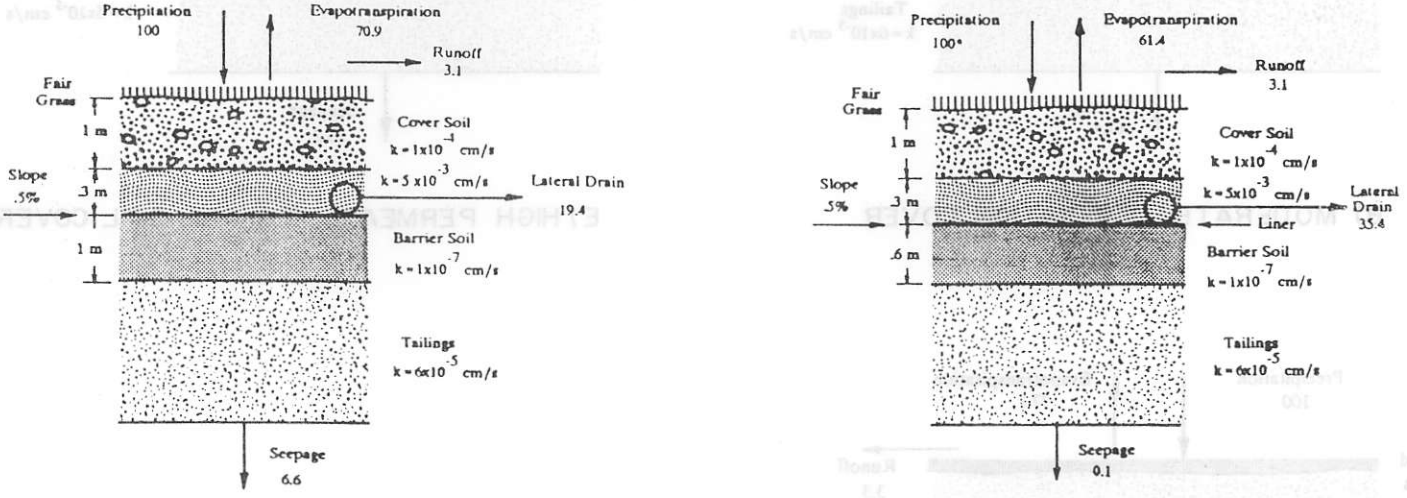
NOTE: (*) Numbers are percent of total annual precipitation

FIGURE 7.4-15a
EFFECT OF COVER TYPE ON INFILTRATION RATE
(After Steffen Robertson & Kirsten, 1988b)



F) RIPRAP EROSION RESISTANT COVER

H) SOIL BARRIER WITH LATERAL DRAIN AND RIPRAP COVER



G) SOIL BARRIER WITH LATERAL DRAIN AND SOIL COVER

I) SOIL BARRIER WITH SYNTHETIC MEMBRANE LINER, LATERAL DRAIN AND SOIL COVER

FIGURE 7.4-15b
EFFECT OF COVER TYPE ON INFILTRATION RATE
(After Steffen Robertson & Kirsten, 1988b)

seepage barrier. This cap reduces surface runoff to 3.1% of the annual rainfall. The lateral drain intercepts 19.4% while evapotranspiration accounts for 70.9% leaving 6.6% as seepage. This is a 62% reduction in the total seepage as compared with bare or vegetated tailings. The major finding is that these layers are effective but not 100% efficient in limiting seepage. At 6.6% infiltration this represents approximately 60 mm of precipitation or 60,000 m^3/yr from a 100 ha disposal site at the Elliot Lake project.

Rock/gravel surface layers have a major effect, increasing the infiltration. With a pervious surface zone, the lateral drains become more efficient reducing the seepage rates.

An engineered cap with a synthetic membrane liner is by far the most effective infiltration barrier. The seepage rates predicted for an engineered cap with a liner that was 99 percent efficient are 0.1% of the annual rainfall. The life of the liner, however, needs to be considered. Synthetic liners are discussed in more detail in Section 7.4.4.

Although theoretical simulations are useful in comparing alternative cover types, the true effectiveness of covers in controlling acid mine drainage can only be determined from monitoring the performance of actual covers in the field. Unfortunately, monitoring results are limited.

The best documented case of a soil cover in use on an actual mine waste dump is that of the Rum Jungle uranium and copper mine in Australia (NTDME, 1986; Bennett et al, 1988; Ritchie and Harries, 1987; Gibson and Portelis, 1988; Verhoeven, 1988). Composite covers were placed on three acid-generating overburden heaps. The top surface covers consisted of a 225 mm compacted clay layer, overlain by a 250 mm sandy clay loam retention zone layer, overlain by a 150 mm gravelly sand erosion layer. Rehabilitation of the heaps also included reshaping their surfaces and providing surface drainage systems. A typical cross section of the rehabilitated heaps is shown in Figure 7.4-16. The largest of the three heaps, Whites Heap was covered in 1983/84. The other two were covered in 1984/85.

Measurements of oxygen concentrations in the pore gas in the heaps show a marked reduction in oxygen concentrations after installation of the compacted clay cover (Bennett et al, 1988). Although measurements indicate that the transition rate of gas through the seal has increased since its initial placement, due to dessication cracking in the dry season, the oxygen concentrations in the heaps are still much less than they were before rehabilitation (NTDME, 1986). The effect of this reduction of oxygen concentration on oxidation rates has not been quantified (NTDME, 1986).

Pre and post rehabilitation measurements on and near the heaps indicate that the cover has provided some reduction in infiltration (NTDME, 1986). However, the amount of that reduction remains in question. Estimates based on lysimiter measurements indicate reductions greater than 90%, while others based on groundwater estimates indicate only a 50% reduction (NTDME, 1986).

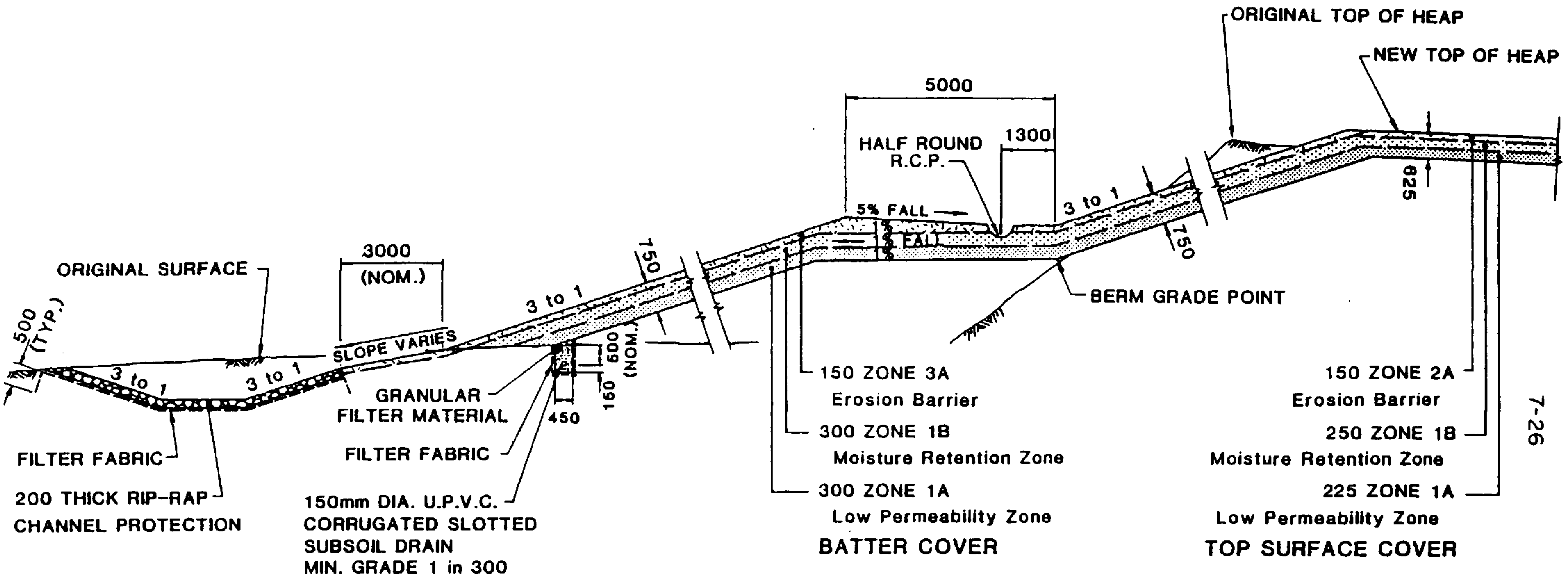


FIGURE 7.4-16
TYPICAL CROSS-SECTION OF REHABILIATION HEAP
SHOWING COVER DETAILS AT RUM JUNGLE
(after Verhoeven, 1988)

The final measure of the effectiveness of the covers, though, is reduction of metals loads in the local river system. Precipitation, flow and metal load data are summarized in Tables 7.4-2 and 7.4-3. The four wet seasons during and after rehabilitation (1984/85) have all been below average rainfall and well below average runoff (Verhoeven, 1988). The reductions in post-reclamation metal loads determined by the Northern Territory Department of Mines and Energy (NTDME, 1986) are indicated in Table 7.4-4. Although samples taken from the East Finniss River show large reductions in metals loading it is unclear whether these reductions are due to covering the heaps, or due to variations in precipitation rates, i.e., it is not clear whether the reduction in the loads is due to reduced acid generation and migration resulting from cover placement or merely due to reduced migration resulting from low precipitation in the years following cover placement.

TABLE 7.4-2

EAST BRANCH OF THE FINNISS RIVER REVISED POLLUTION LOADING VALUES

(after NTDME, 1986)

Season	1971/72	1972/73	1973/74
Rainfall (mm)	1542	1545	2000
Total Flow $m^3 \times 10^6$	31	22	69
Metal Load (t)			
Copper	77	67	106
Manganese	84	77	87
Zinc	24	22	30

TABLE 7.4-3

SUMMARY OF MONITORING RESULTS FOR THE EAST BRANCH OF THE FINNIS RIVER

(after NTDME, 1986)

Season	1982/83	1983/84	1984/85	1985/86
Rainfall (mm)	1121	1704	1112	910
Total Flow $m^3 \times 10^6$	9.5	48	11.7	11.4
Metal Load (t)				
Copper	23	28	9	4
Manganese	6	21	7	8
Zinc	5	9	4	3

TABLE 7.4-4

PERCENTAGE REDUCTION IN POLLUTION OF THE EAST BRANCH OF THE FINNISS RIVER

(after NTDME, 1986)

Season	1983/84 (Stage 1 & part Stage 2 complete)	1984/85 (Stage 3 complete)	1985/86	Target
Metal				
Copper	70%	80%	91%	70%
Manganese	76%	88%	86%	56%
Zinc	67%	73%	79%	70%

One interpretation of the results of surface water monitoring of the East Finniss River indicates a decline in annual copper loads since placement of the waste dump covers (Verhoeven, 1988), as shown in Figure 7.4-17. However, a separate evaluation of rainfall, flow rates and metal loads appears to show no improvement in metal loads for the 1984/85 and 1985/86 seasons (Steffen Robertson and Kirsten, 1988(b)). The analysis by Verhoeven is qualified by the fact that the rehabilitation works have as yet not been subjected to an average or above average wet season, and that environmental response may be slow. These analyses serve to indicate that more monitoring results are required to conclusively assess the effectiveness of the rehabilitation measures.

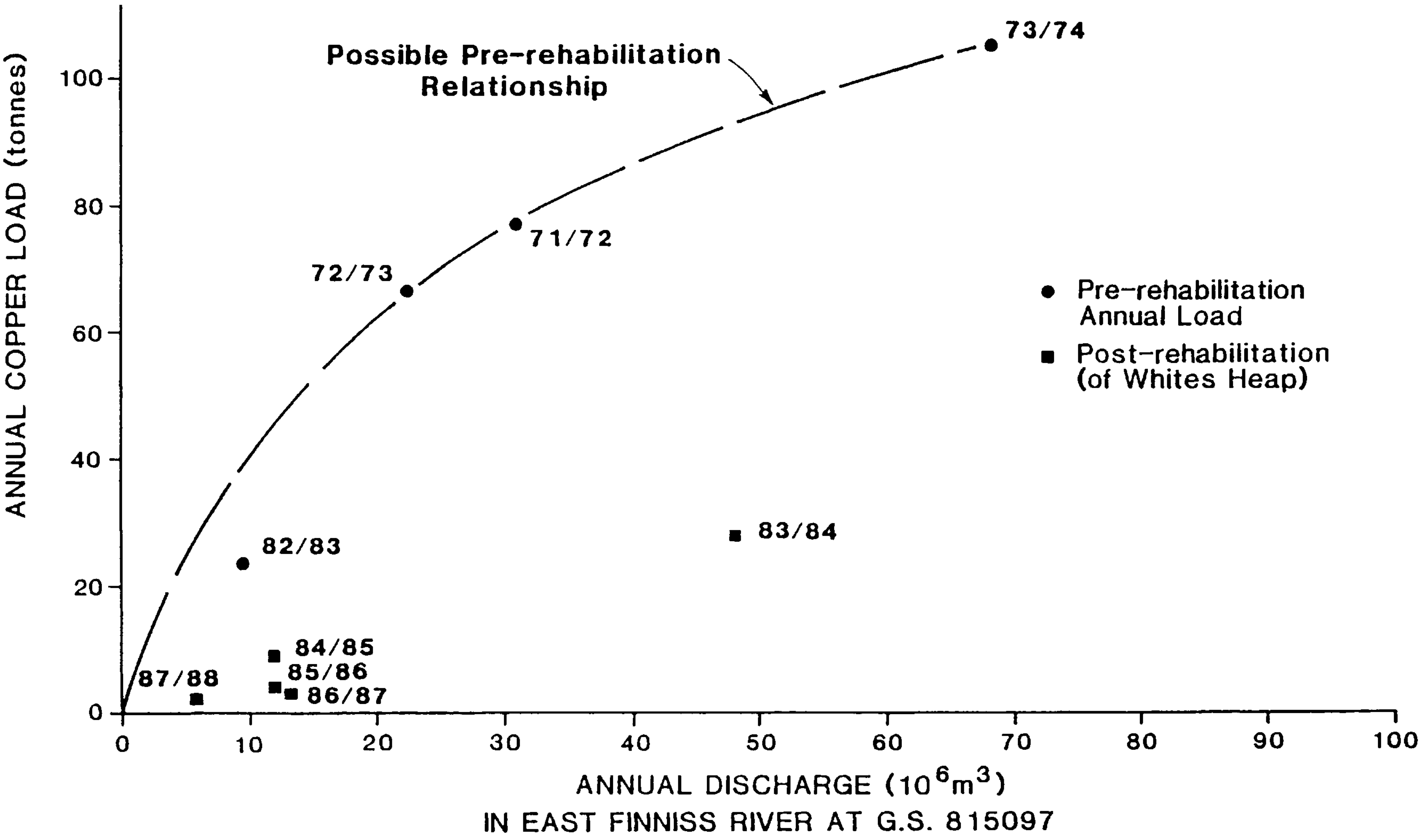


FIGURE 7.4-17
COPPER LOAD VS. DISCHARGE IN EAST FINNISS RIVER
(after Verhoeven, 1988)

Measurements of groundwater quality criteria beneath rehabilitated dumps at Rum Jungle indicate that the groundwater quality has not changed significantly during the four years since cover placement (Gibson and Pantelis, 1988). It would appear that there is a store of contaminants in the groundwater and within the dump. The response of downstream surface-water quality to cover placement may well be affected by the release of these stored contaminants. On-going monitoring is essential until trends in surface and ground water quality are established.

Results from the AMD Task Force questionnaire indicated that soil covers have been placed at three mines in B.C. and are the preposed control measures at a number of other mines (Steffen Robertson and Kirsten, 1988a). While the placed covers appear to be effective to date, the period of monitoring is considered too short for valid conclusions. A till cover is at present being placed over waste dumps at the abandoned Mt. Washington mine on Vancouver Island. Instrumentation will be installed to monitor oxygen levels, temperature and infiltration of precipitation. Construction of the cover is expected to be complete at the end of summer, 1989. Monitoring of this facility in future years will provide valuable information for evaluating the effectiveness of such oovers.

7.4.3 Synthetic Covers

The use of synthetic membranes for liners for tailings impoundments has been reviewed by Golder Associates, (1984).

7.4.3.1 GEOMEMBRANES

Flexible membrane liners are commonly referred to as geomembranes. Common types are:

i) Polyethelene (P.E.).
ii) High density polyethylene (HDPE).
iii) Chlorinated polyethelene (CPE).
iv) Chlorosulphonated polyethylene (CSPE commonly known by the Dupont trade mark - HYPALON).
v) Polyvinyl chloride (PVC).
vi) Ethylene propylene diene monomer (EPDM).
vii) Butyl rubber.

Occasionally neoprene and polyurethane are also used. Collectively, synthetic membranes display a number of advantages and disadvantages which may be summarized as follows:

Advantages:

- i) They can contain a wide variety of fluids with minimum seepage due to low reported permeabilities of typically 1 x 10^{-1}0 cm/sec or less.
- ii) They have a relatively high resistance to chemical and bacterial deterioration.
- iii) They are readily installed for many applications.
- iv) They are relatively economical to install and maintain.

Disadvantages:

- i) They are relatively vulnerable to attack from ozone and ultra-violet light.
- ii) They have limited ability to withstand stress from heavy machinery.
- iii) They have not been in service long enough to evaluate long term performance.
- iv) They are comparatively susceptible to laceration, abrasion and puncture.
- v) Some materials are prone to cracking and creasing at low temperatures or stretching and distorting at high temperatures.
- vi) Although readily installed there are often difficulties associated with seaming of the material.

Polymeric membranes offer wide ranging chemical resistance and may be readily inspected. However, they are susceptible to damage during installation largely due to improper subgrade preparation and vehicular traffic. They require very careful installation and their performance is dependent on careful and successful field seaming. Field seaming is, in general, a detailed and sensitive operation. Weather, including temperature and precipitation, is generally the governing factor. In this regard, the elastomeric liners namely, Butyl, Polychloroprene and EPDM would appear to present the most problems in field seaming. Of the remaining liner types considered, successful field seaming has been demonstrated with HDPE, PE, CSPE, CPE and PVC. It is noted, however, that there are serious concerns about the long-term weatherability of PVC and PE.

Proper subgrade preparation and construction is crucial for a successful liner installation and would typically consist of excavation of compressible materials, sterilization of the subgrade, removal of all roots, sticks, stones and debris, grading and proof-rolling, and installation of the sand cushion, liner and soil cover. Installation of the liner and field seaming should be carried out by approved installers meticulously following liner supplier instructions. Soil cover is desirable but will require liner inclinations flatter than about 3 horizontal to 1 vertical. This is a severe limitation when applied to waste dumps.

With the exception of polyurethane the base polymeric resins and asphalt show promise for long term resistance to the major anticipated constituents of acid mine drainage. However, CPE may be affected by weak sulphuric acid solutions.

Thin flexible membrane liners are susceptible to overstressing by strains associated with large differential deformations in the subgrade. It may be necessary to subexcavate and replace compressible materials encountered over the subgrade prior to liner installation. Similar concerns exist for liners placed on slopes and where there is a potential for excess hydrostatic or gas pressure buildup beneath the liner.

Seepage through liners is primarily through liner defects. A rational approach to evaluating apparent or field liner permeabilities is through detailed monitoring of existing installations. An experimental study to evaluate the rate of leakage of liquids through a flawed liner has been carried out at the Texas A & M University (Jayawickrama, et al, 1988). The results of the study show that the type and thickness of the membrane liner have a relatively small influence on leak rates while the low permeability subbase is important in restricting flow through a flawed liner.

Estimates of liner release rates were undertaken by Golder Associates, 1984, assuming permeable subsurface conditions (permeability greater than 1×10^{-4} cm/sec) and tailings permeability of 1×10^{-5} cm/sec. The analyses indicate that an asphaltic membrane would reduce seepage to about 50 percent of an unlined basin for a field liner permeability of 1×10^{-8} cm/sec. Polymeric liners with an effective permeability of 1×10^{-10} cm/sec would reduce seepage to less than 10 per cent of an unlined basin.

In West Virginia, a polyvinyl chloride liner was used to cover a 45 acre backfilled site to prevent seepage into acid producing materials (Caruccio and Geidel, 1983). Results showed substantial decreases in flow and acidity from associated seeps (Caruccio and Geidel, 1986).

Used as the barrier layer in combination with soil material layers in a complex cover, geomembranes should prove to be very effective in limiting oxygen and water transport.

7.4.3.2 ASPHALTIC AND SPRAY ON SEALS

Asphaltic and spray on surface sealants can be applied to the surface of the waste to form a barrier to infiltration and oxygen diffusion. A number of products are available, including:

- Alkyd
- Asphalt
- Concrete
- Epoxy
- Polyester
- Polysulfide
- Polyurethane
- Silicone
- Synthetic Rubber
- Thermoplastic Molten Sulphur
- Vinyl

These materials have, in general, been developed for applications such as caulking sealants, soil stabilizers, waterproof barriers, and corrosion protective coatings. Their application to date in mine waste covers is limited.

Surface sealants can be formulated to produce either flexible or rigid linings or covers. Surface sealants can be installed with three basic techniques:

i) In-Situ Chemical Cure

The materials chemically cure or harden after being applied to the surface. These materials usually involve more than one specific chemical.

ii) Heat Application

Materials which are solid in the desired operating temperature range are applied at elevated temperatures to improve ease of application.

iii) Surface Drying

The material is formulated in a water emulsion or diluted in a solvent carrier for application. The carrier evaporates leaving a solid coating.

Combinations of the above techniques are also feasible in many cases. The object is to prepare the material for ease of application usually with conventional spraying equipment. The actual technique for application is a function of the specific material.

The primary advantages and disadvantages of surface sealants are:

Advantages

a) Either sufficient flexibility to conform with or sufficient strength to support the design load (pedestrian or vehicle traffic for example).

b) Good weatherability and service life.

c) Compatibility with the stored product.

d) Immunity to biological attack.

e) Sufficient puncture and abrasion resistance.

f) Capability of being placed with minimal defects.

g) Easily repairable.

h) Ease of application and produces an integral liner with no joints.

Disadvantages

a) Relatively difficult to regulate the rate of application and thus the thickness and uniformity of the sealant.

b) As a class these materials are relatively expensive. The high initial cost versus relative ease of application for the spray-ons should be considered for specific applications.

Pacific Northwest Laboratories have investigated the effectiveness of various asphaltic seals for use as radon barriers on uranium tailings deposit (Baker, Hartley and Freeman, 1984). Types of seals tested included sprayed-on asphalt emulsion, in-situ addition of asphalt emulsion, hot rubberized asphalt seal, and asphalt emulsion admixes. They found that, though all applications significantly reduced radon flux, admix seals such as those applied with a cold mix paver appeared to be significantly more stable than the other seals. Admix seals could not be applied to surfaces sloping more than 12%.

Test results on radon barrier systems using an asphalt emulsion admix seal are described by Hartley et al (1982).

The asphalt emulsion system consists of an 8 cm thick asphalt emulsion/aggregate admix seal that is covered with ~0.6 m of overburden. The admix seal, which forms the radon-impermeable diffusion barrier, is a mixture of cationic asphalt emulsion and an aggregate such as concrete sand. When cured, it contains ~ 22 wt% asphalt and less than 1 wt% water. The overburden stabilizes the admix seal and protects it from UV exposure, rain, extreme temperatures, and sudden temperature changes. Filter/drainage layers would have to be added above and below the seal to prevent degradation due to frost action in Canadian climates.

Tests performed at Pacific Northwest Laboratories showed that asphalt seals are very effective in reducing radon flux. Asphalt seals were shown to have very low diffusion coefficients to radon (see Table 7.4-5); and, field tests showed that asphalt seals could reduce radon flux by more than 99%. Asphalt seals should, therefore, be very effective in reducing oxygen diffusion as well. They also provide very good barriers to infiltration, their hydraulic conductivities being around 10^{-20} m/s (Nolan Davis and Associates, 1987).

TABLE 7.4-5

EFFECTIVE DIFFUSION COEFFICIENTS FOR RADON THROUGH ASPHALT SEALS

(after Hartley et al, 1982)

Seal Description	De, $cm^2/s \times 10^{-6}$
Asphalt Cement	0.44 to 1.5
Rubberized Asphalt	0.12 to 0.15
Laboratory prepared Asphalt Emulsion Admix Seals	1.6 to 31
Grand Junction Field Test Asphalt Emulsion Admix Seal Core Samples	2.2 to 10
Typical Earthen Cover	10,000 to 20,000

The concern with asphalt seal is their long-term stability. Hartley et al (1982) stated that aqueous leaching and oxidation should not be expected to degrade the seal, and that the mechanical properties of the asphalt seal are more important in assessing its long-term stability. Further testing of the long-term mechanical properties of asphalt seals will be required.

7.4.3.3 GEOPOLYMERS

Geopolymer is the term given to a compound of minerals, principally containing silica, phosphate and oxygen that bond to form a ceramic type productl The suitability of using this product as a control measure for acid generation is currently being investigated in Canada by CANMET. It is anticipated that geopolymers may be mixed with tailings to form a solid mass preventing oxygen and/or water access to sulphides. A possible alternative is to mix the geopolymer with soil or other material and apply this as a cover to the waste. The behaviour of geopolymers when mixed with different waste materials is not fully known at this stage, nor the resistance to natural processes such as freezing and thawing. Geopolymers are still in the development stage and require extensive research to establish their suitability as a control measure.

7.4.3.4 SHOTCRETE

Shotcrete is the name given to concrete pneumatically delivered through a hose and applied onto a surface at high velocity. Shotcrete may be effective in the control of acid generation when applied as a cover to certain wastes. Ths advantage of shotcrete is that it can be applied to steep rock slopes or other surfaces which may be difficult to cover using other methods.

The effective use of shotcrete as a cover le dependent on the stability of the underlying material. This method has been used very successfully on rock faces and on compacted materials. However, if the material to which the shotcrete is applied undergoes consolidation or settlement causing relative displacement at the surface, cracking of unreinforced shotcrete will occur. Once the shotcrete liner has suffered cracking, the effectiveness of the cover is lost. Experience has shown that displacement of uncompacted waste dumps often occurs and for this reasons unreinforced shotcrete is not appropriate as a cover to these materials. High temperature induced stresses in a shotcrete cover on the surface of a waste facility may also result in cracking.

The resistance of shotcrete to cracking may be increased by providing reinforcing. Conventional steel mesh reinforcing is expensive, difficult to handle and subject to corrosion in the long-term. Steel fibre reinforcement is easier to apply, however, is also vulnerable to corrosion. A method of reinforcement using high volume polypropylene fibre reinforcement, which is corrosive resistant and relatively flexible, shows promise (Morgan et al, 1989). A benefit of the fibre type reinforcement is that it reduces crack widths in the shotcrete. This gives the shotcrete cover the facility to accommodate larger movements than mesh reinforced shotcrete.

7.4.4 Placement of Covers

The establishment of covers on mine waste and tailings is complicated by the difficulties of access, trafficability and stability of the surfaces onto which the cover is to be placed. These difficulties often render a particular cover type impractical or prohibitively expensive. Placement of some cover types requires access of wheeled vehicles working on fairly flat surfaces (asphalt covers), others require careful bed preparation and moderate slopes (synthetic membranes) and others a firm surface against which to compact (clay layers).

7.4.4.1 WASTE DUMP SURFACES

Rock waste surfaces are conveniently subdivided into the dump surface and the dump slopes; with different conditions applying to each.

Dump Surface

The upper surface of a rock waste dump is usually readily accessible, trafficable and nearly flat. The placement of any type of surface cover, except a water cover, is usually not difficult.

During dump development the material on the upper surface of dumps placed by trucking is often broken down and compacted under the wheel traffic of the dump trucks. This results in a fairly compact lower permeability upper surface. This surface reduces infiltration and ponding is often experienced on such surfaces. Despite the initial coarse nature of the material in such dumps it may be necessary to install a suction breaking layer to prevent downward suction on low permeability cover layers.

The dumps are subject to long term consolidation and settlement under the self weight of the dump and as the dump rock weathers. These settlements are large (a few percent of the dump height), and uneven, reflecting the natural variation of the waste rock and dumping procedures. Differential settlements result in disruption of the drainage pattern on the dump surface and cracking of cover materials. Settlement and crack patterns are often such that drainage is towards cracks resulting in considerably increased infiltration.

Dump Slopes

Dump slopes are usually placed at their angle of repose. At this angle slopes are inaccessible, untrafficable and marginally stable. Cover placement on such steep slopes is essentially impractical. Crest dumping of cover materials has been attempted at some sites creating uncompacted (permeable) uneven covers of questionable stability.

For dump slopes to bs accessible it is necessary to first reslope. At a slope of 3 horizontal to 1 vertical (3:1), the slopes are trafficable by tracked vehicles and it is possible to place soil type cover materials. At this slope it is possible to also place synthetic membrane liners but the stability of cever layers over such membranes is questionable. The cost of resloping large dumps to 3:1 is very large, requiring large expenditures of dozer time, unless the resloping had been planned for and the dumps constructed with a staggered dump slope.

At its natural angle of repose a slope is just stable. Weathering of the dump materials, or increases in the water table with time, may result in slope failure (HMSO, 1967, Campbell and Shaw, 1978, Bishop, 1973). Such slope failure may not be catastrophic but take the form of slope creep. Either will result in the disruption of any cover layers.

Dump slopes are subject to the same concerns regarding differential settlement as are the dump surfaces.

Erosion on the steep dump slopes is a major long term concern. This is considered in greater detail in Chapter 9.0.

7.4.5.2 TAILINGS SURFACES

The access, trafficability and stability of a tailings surface varies tremendously, depending on the nature and age of the tailings deposit and location on the impoundment surface. The effects of placement conditions on the properties of tailings deposits has been reviewed by Robertson (1987). The difficulties of placing covers on tailings and the deformations of tailings surfaces are reviewed by Steffen, Robertson and Kirsten (1986a).

Wet unconsolidated tailings always represent difficult access conditions. Access improves as the tailings are drained and consolidate. Where tailings have been spigotted onto beaches, the sand fraction is deposited nearest the spigots and drain more freely than the slimes which accumulate near and in the pond. Drained sandy beaches may be trafficable within days of deposition while pond areas may never achieve this condition. Thus it is possible to place and compact covers on the beach areas with little preparation. In the pond areas it may be necessary to apply drainage measures to remove free and near surface water, and to use geofabrics on the slimes followed by thin layers of the cover. These techniques have been successful in placing a cover over slimes which could not be walked on at the start of cover placement (Steffen Robertson and Kirsten, 1983). Covers may also be placed during winter when freezing conditions allow access, as was done for the cover placement over wet tailings at the Beaverlodge mine (Steffen Robertson and Kirsten, 1983; Eldorado Resources, 1986).

To prevent capillary suctions in covers it may be necessary to utilize capillary barriers over tailings fines and slimes as discussed in Section 7.7.

Prior to placing covers on tailings it is necessary to first develop a tailings surface which has an adequate slope and drainage pattern. Requirements are reviewed by Steffen, Robertson and Kirsten (1987). Much can be done to achieve such slopes by adopting an appropriate tailings placement and management method (Robertson, 1987). Reshaping of tailings surfaces after closure, with earth moving equipment, may be difficult and prohibitively costly. Trafficability of dewatered tailings are reviewed by Robertson et al, (1982).

After closure, tailings continue to consolidate and settle as a result of dissipation of pore pressures and thawing of included ice. The effects have been reviewed for Canadian tailings impoundments by Steffen, Robertson and Kirsten, (1986a, 1987). These settlements can be a substantial portion of the total tailings depth and result is disruption of the drainage pattern, leading to extensive ponding on the tailings surface, and cracking of covers.

7.4.5.3 ROCK SURFACES

The placement of covers on steep rock surfaces, such as pit walls, poses a particular problem. Two approaches can be used. The first requires the construction of thick self-supporting covers. The second requires adherence of the cover to the rock face and relies on the rock face to support the cover. The use of gunite or shotcrete methods is appropriate for the second. Both asphalt and concrete materials can be considered. Because of the corrosive nature of ARD to cement and steel, the use of synthetic fibres and silica fume concrete is appropriate.

7.5 WASTE ROCK AND TAILINGS PLACEMENT METHODS

7.5.1 Waste Rock Dumps

Control of ARD migration in waste rock dumps can be assisted by engineered placement methods such as cellular dump construction, compacting, mixing with low permeability material, etc.

Cellular dump construction, when used in conjunction with a cover layer, can significantly reduce the area exposed to precipitation. Cellular construction may utilize a layer construction or lateral cell construction. This method is a short term control as it applies only to the construction period.

Control of ARD migration by compacting on mixing with low permeability material are both intended to reduce the bulk permeability of the waste rock dump which will reduce infiltration. They can be used together or separately.

Compacting of waste rock will require dump construction in thin layers and is only suitable for soft rocks. This approach will also reduce settlement and increase dump stability. The result is long term integrity and improved cover performance. Compacting will be very expensive.

Mixing low permeability material into the waste rock will reduce infiltration. A non-acid generating waste material, or preferably a net alkaline waste, can be mixed during dumping. Dump stability will have to be addressed. This approach should provide good long term reduction of Infiltration.

7.5.2 Tailings

Control of ARD migration in tailings impoundments is limited to methods which reduce the permeability of the tailings with density control. These methods include pre-disposal thickening, sub-areal deposition, flocculant addition, and installation of dewatering system such as wick drains. It may be possible on some sites to reduce tailings permeability with the addition of clay to the tailings line. All of these methods will provide long term control of ARD migration. Methods which increase the density may be required for stability in earthquake prone regions.

7.6 CONCLUSIONS

1. Control of ARD migration will be best achieved by integration of control methods including:

    - diversion of surface water
    - groundwater interception
    - infiltration control
    - waste placement methods

2. Diversion of surface water is best achieved during operation (short term) by means of diversion ditches or berms. In the long term, site selection to minimize contact with surface water runoff should be considered if possible. If necessary, ditches, berms and other structures may be used in the long term, however, a certain level of inspection and maintenance will be required.

3. Interception of groundwater by means of wells and pumps may be suitable in the short term only. Impermeable cut-off walls and gravity drains may be suitable in the long term but will require on-going monitoring and maintenance.

4. Infiltration control is essential for controlling ARD migration. This is best achieved by means of soil, synthetic materials, or a combination of these. Synthetic membrane liners are most suitable in the short term to cover, for example, ore stockpiles. The design of soil covers must consider the degree of infiltration control required, the requirements for revegetation, long-term disruptive forces and maintenance requirements. All types of covers require some form of long-term monitoring and maintenance.

5. Methods of placing waste rock, spoil or tailings to minimize infiltration should be considered in conjunction with other control methods.

6. The methods of control of ARD migration are sensitive to both the nature of the site and the duration for which control is required. Consideration of all site parameters is critical to selecting the optimum combination of methods.

CHAPTER 8: COLLECTION AND TREATMENT OF ARD

8.1 INTRODUCTION

The third level of control, after control of acid generation and ARD migration, is to collect and treat contaminated drainage. In some cases at existing facilities this is the only practical option available. The collection and treatment of ARD has been the most widely applied abatement measure to date. Treatment measures can be either active systems which require continuous operation such as a chemical treatment plant or passive systems which are intended to function without interventien by man, such as wetlands. Chemical treatment involves technology which is well established and is working effectively at a number of mines. According to the AMD Task Force Questionnaire, a total of four mines are practicing chemical treatment in B.C.

The main disadvantages of chemical treatment are; it requires continuous operation and maintenance, the relatively high risk of equipment or power failure makes back-up or contingency measures necessary, and the need to dispose of large volumes of sludge. Chemical treatment, while it offers a secure short-term method of achieving environmental protection, may not offer a cost effective long-term solution.

8.2 COLLECTION SYSTEMS

Collection requires collection of both surface waters and groundwater contaminated by ARD. The occurrence of ARD in the form of surface flows is more visible and in general is more harmful to the aquatic environment than groundwater contamination. Collection of surface flows is usually fairly readily achieved by means of surface ditches. The collection of subsurface flows requires the installation of collection trenches, 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. The design of the appropriate collection system is extremely site specific and depends on location of contaminated flow, topography, precipitation and runoff properties, and the geohydrology of the site.

8.3 CHEMICAL TREATMENT

8.3.1 Process Chemistry

The objectivo of the acid mine drainage treatment process is to eliminate acidity, precipitate heavy metals and remove any deleterious substances such as suspended solids, arsenate and antimonate.

Acidity is eliminated by neutralization of the acid stream with an alkali reactant or combination of alkali reactants offering the best economics. Heavy metal ions hydrolyze and precipitate as hydroxides in the resulting neutral solution due to their limited solubilities. Anions such as arsenate and antimonate form insoluble compounds at neutral pH with many of the heavy metals (notably iron) and thus may be removed simultaneously. In circumstances where insufficient heavy metal is present in solution to accomplish complete anion removal, addition of metals such as iron may be necessary prior to neutralization.

In circumstances, where precipitation during neutralization is inadequate, other secondary methods such as sulfide precipitation or ion exchange may be considered. In some cases, the concentration of dissolved heavy metals may be high enough to warrant processing the solution for the metal recovery. Solvent extraction, ion exchange, or precipitation may be the first step used in these types of schemes. Such systems are in the realm of hydrometallurgical process technology, and outside the scope of this section.

8.3.1.1 NEUTRALIZATION

Sulfuric acid in acidic mine drainage originates from oxidation of sulfides. The primary choices for the basic neutralization reagent will be ground limestone (calcium carbonate, $CaCO_3$) and slaked lime (calcium hydroxide, $Ca(OH)_2$). The reactions for these reagents are (Masters, et al, 1980):

$$H_2SO_4 + CaCO_3 + H_2O \text{ --> } CaSO_4.2H_2O + CO_2 \tag{8.1}$$

$$H_2SO_4 + Ca(OH)_2 \text{ --> } CaSO_4.2H_2O \tag{8.2}$$

The primary product of both of these reactions is gypsum ($CaSO_4.2H_2O$), produced largely as a precipitate. Gypsum, however, is prone to supersaturation and may persist in solution at levels above its nominal solubility. Gypsum commonly forms scale when supersaturation is relieved at flow irregularities which provide seeding sites in piping systems. Scaling problems can lead to significant maintenance expenditures.

Dolomitic lime and limestone are similar to conventional lime and limestone except they contain a proportion of magnesium in place of calcium. The neutralization reactions with dolomitic lime or limestone produce some soluble magnesium sulfate in addition to the gypsum shown in reactions 8.1 and 8.2. This results in some added dissolved solids loading.

Selection of the type of lime or limestone is generally governed by price and availability, while the selection of lime versus limestone is not as straightforward. Limestone, while a cheaper reagent, has some limitations. Gypsum reaction products tend to blind the limestone particle surface which reduces neutralization efficiency and increases consumption. The use of finely ground limestone and/or strong agitation is necessary to improve utilization. CO_2 produced via limestone neutralization can interfere in the subsequent gravity liquid-solid separation step. CO_2 bubbles can be entrapped in the precipitates causing some of the flocs to float during the settling process.

In any case, it is not practical to achieve complete neutralization using limestone, even though it has an equilibrium pH above 7 (Garrels and Christ, 1965). The kinetics of the neutralization reaction with limestone near neutral pH are slow. Neutralization to the pH 8-9 range, necessary to precipitate metals such as zinc will require a stronger base, such as $Ca(OH)_2$.

Other bases that may be considered for neutralization include (Baker, 1974):

1. Caustic Soda (Sodium hydroxide): usually rejected because of reagent cost. Capital costs for feeding and handling are less than for lime. Contributes dissolved solids to the effluent due to the solubility of most sodium salts, but produces less sludge as a result. Minimal precipitate formation may limit effectiveness of any co-precipitation/surface adsorption scavenging mechanisms. May be relatively ineffective for removing anions if suitable cations to precipitate these are not already present. May be used in smaller installations where chemical costs are less significant.

2. Soda Ash (Sodium carbonate): generally has the same characteristics for neutralization as caustic soda. In addition, it may evolve CO_2 in the reaction, with possible detrimental effects on settling/clarification. May be used in particular cases (e.g. cadmium and nickel precipitation) where it provides a better chemical reaction, presumably due to carbonate precipitation rather than hydroxide precipitation (Lanouette, 1977).

3. Ammonia: characterized by high reactivity, high solubility of product salts, and high neutralization capacity on weight basis. Contribution of available nitrogen to the effluent and the toxicity of the ammonium ion usually preclude its use. In excess, ammonia is a powerful complexing agent for some heavy metals and may in fact hinder precipitation.

4. Sodium sulfide: used where the lower solubility product of heavy metal sulfides will provide more effective removal (Lanouette, 1977). Less convenient to use than other bases, as excess sulfide in solution will form toxic H_2S, requiring further treatment. Certain heavy metals such as mercury, tin, and silver will be resolubilized in an excess of sulfide due to formation of sulfide complexes, so that correct dosage is critical (Anderson and Weiss, 1973). Sulfide sludges may oxidize to form acid, thereby re-solubilizing metals and limiting long term stability.

5. Mill tailings, flash, etc.: can be attractive due to cost. Testing is essential to ensure that their use does not introduce undesirable components such as cyanide. Sludge volumes generated by tailings or fly ash may be much greater than with more concentrated bases, with potential impact on treatment plant and sludge storage facility sizing. These materials may not have sufficient basicity to achieve complete neutralization, but can lower overall reagent costs by reducing the consumption of more expensive reagents such as hydrated lime.

8.3.1.2 PRECIPITATION

Most metals are precipitated as their respective hydroxides or hydrated metal oxides, while the associated sulfate is precipitated as gypsum (Masters et al, 1980). Depending on the history of the solution, iron may be present as both ferrous and ferric ions. During neutralization, the slurry is aerated to oxidize ferrous iron to ferric. The major portion of this oxidation occurs after the ferrous iron has already been precipitated as a hydroxide.

The anionic species of arsenic, antimony, phosphorus, vanadium, titanium, and silicon are precipitated during neutralization as calcium or heavy metal compounds, usually ferric compounds. During initial neutralization, a significant portion of any arsenate or antimonate will be precipitated with the heavy metals present. As neutralization proceeds into the alkaline range, these compounds may be converted to calcium arsenate and antimonate. Ferric iron compounds of arsenic and antimony are more stable, however, and the presence of ferric iron is desirable to achieve low contaminant levels in solution and produce stable sludges.

Equations for some of these reactions are as follows:

Precipitation of ferric iron:

$$Fe_2(SO_4)_3 + 3\ Ca(OH)_2 + 6\ H_2O \dashrightarrow 2\ Fe(OH)_3 + 3\ CaSO_4.2H_2O \qquad (8.3)$$

Precipitation and oxidation of ferrous iron:

$$2\ FeSO_4 + 2\ Ca(OH)_2 + 1/2\ O_2 + 5\ H_2O \dashrightarrow 2\ Fe(OH)_3 + 2\ CaSO_4.2H_2O \qquad (8.4)$$

Precipitation of arsenate and antimonate with ferric iron:

$$2\ H_3AsO_4 + Fe_2(SO_4)_3 + 3\ Ca(OH)_2 \dashrightarrow 2\ FeAsO_4.2H_2O + 3\ CaSO_4.2H_2O \qquad (8.5)$$

$$2\ H_3SbO_4 + Fe_2(SO_4)_3 + 3\ Ca(OH)_2 \dashrightarrow 2\ FeSbO_4.2H_2O + 3\ CaSO_4.2H_2 \qquad (8.6)$$

Precipitation of arsenate with lime:

$$2\ H_3AsO_4 + 3\ Ca(OH)_2 \dashrightarrow Ca_3(AsO_4)_2 + 6\ H_2O \qquad (8.7)$$

Other metals:

$$Al_2(SO_4)_3 + 3\ Ca(OH)_2 + 6\ H_2O \dashrightarrow 2\ Al(OH)_3 + 3\ CaSO_4.2H_2O \qquad (8.8)$$

$$CuSO_4 + Ca(OH)_2 + 2\ H_2O \dashrightarrow Cu(OH)_2 + CaSO_4.2H_2O \qquad (8.9)$$

$$ZnSO_4 + Ca(OH)_2 + 2\ H_2O \dashrightarrow Zn(OH)_2 + CaSO_4.2H_2O \quad (8.10)$$

$$NiSO_4 + Ca(OH)_2 + 2\ H_2O \dashrightarrow Ni(OH)_2 + CaSO_4.2H_2O \quad (8.11)$$

The effectiveness of the precipitation process is limited by the solubilities of the precipitates at the pH of the system. While knowledge of solubility product constants of the precipitate permits calculation of the levels attainable, in many instances the plots of metal ion concentration against pH show a minimum due to the amphoteric properties of some species. In addition, coprecipitation and surface adsorption complicates theoretical prediotion. The kinetics of the neutralization reaction and the presence of colloidal compounds can influence the metal concentration obtained with real solution.

Theoretical equilibrium solubility curves for iron, zinc and copper hydroxides or oxides are presented in Figure 8.3-1 (Stumm and Morgan, 1981).

Each species has an optimum pH (level of neutralization) for its removal. The overall optimum pH for some metals is above acceptable limits for final effluent, therefore pH adjustment may be necessary prior to discharge. Testwork conducted to simulate the process is essential to evaluate process efficiency and establish optimal conditions.

As shown in equations (8.5) and (8.7), arsenic is removed as ferric arsenate (scorodite) or calcium arsenate. Arsenites may also be present but will oxidize to arsenate in the presence of ferric iron. Arsenite precipitates are generally more soluble than their corresponding arsenates (Robins, 1985). However ferric iron will probably be present in the feed, since it is a major product of bacterial pyrite oxidation leading to ARD. If the waste rock or tails have a very high arsenic content, supplemental iron may be necessary to ensure adequate treatment (Lanouette, 1977).

Low dissolved metal levals beyond those achievable by neutralization and hydroxide precipitation can be achieved using sulfides, as illustrated in Table 8.3-1 which presents solubilities of metal hydroxides and sulfides in pure water (Lanouette, 1977).

Systems using sulfide precipitation should be preceded by neutralization to achieve good dosage control. One scheme uses ferrous sulfide as the source of sulfide ion. The sulfide is released only when heavy metals having lower equilibrium constants for their sulfide form are present in solution. At alkaline pH with aeration, the liberated iron will form a hydroxide and precipitate out as well:

$$FeS + Cu^{++} \dashrightarrow CuS + Fe^{++} \quad (8.12)$$

$$Fe^{++} + 1/4\ O_2 + 1/2\ H_2O + 2\ OH^- \dashrightarrow Fe(OH)_3 \quad (8.13)$$

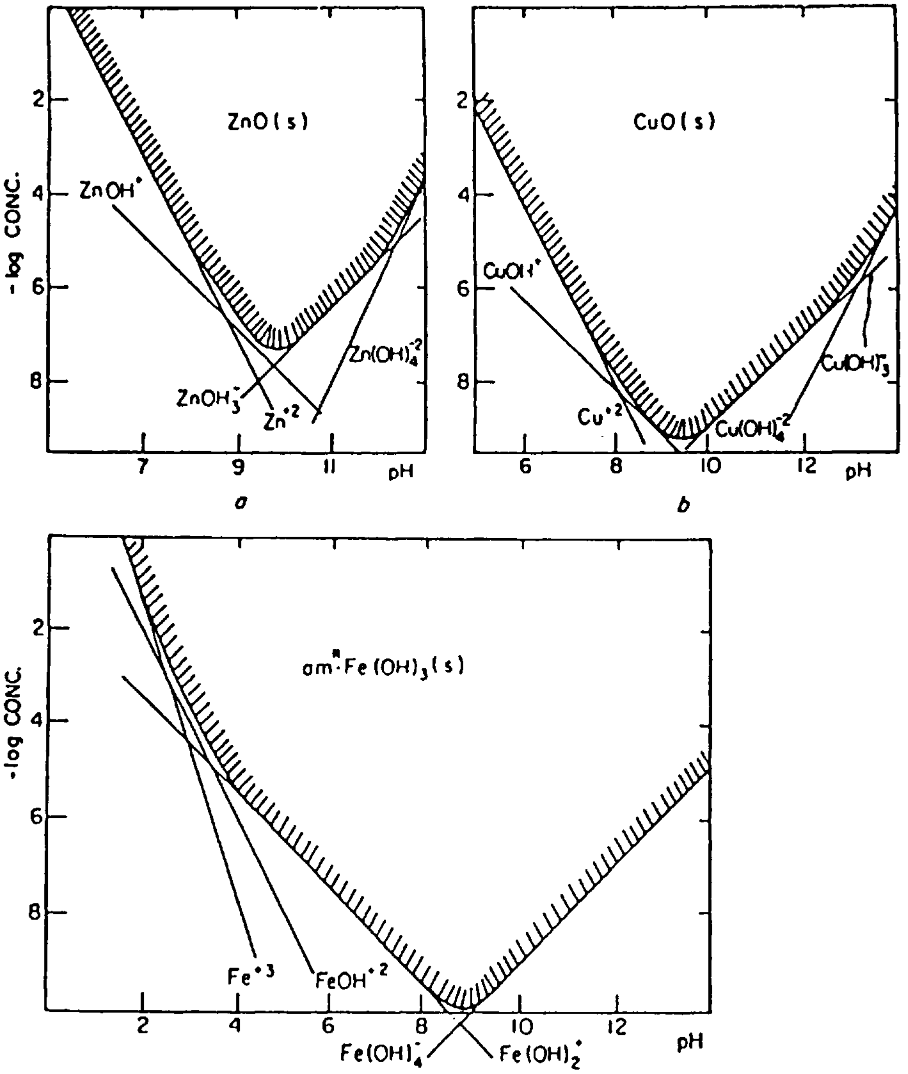


FIGURE 8.3-1
SOLUBILITY OF AMPHOROUS $Fe(OH)_3$,ZnO, & CuO
(After Stumm and Morgan, 1981)

TABLE 8.3-1
SOLUBILITY OF METAL ION, Mg/L
(after Lanouette, 1977)

Metal	As Hydroxide	As Sulfide
Cadmium (Cd^{++})	2.3×10^{-5}	6.7×10^{-10}
Cobalt (Co^{++})	2.2×10^{-1}	$1.0 \times 10^{-8+}$
Copper (Cu^{++})	2.2×10^{-2}	5.8×10^{-18}
Iron (Fe^{++})	8.9×10^{-1}	3.4×10^{-5}
Lead (Pb^{++})	2.1	3.8×10^{-9}
Manganese (Mn^{++})	1.2	2.1×10^{-3}
Mercury (Hg^{++})	3.9×10^{-4}	9.0×10^{-20}
Nickel (Ni^{++})	6.9×10^{-3}	6.9×10^{-8}
Silver (Ag^{++})	13.3	7.4×10^{-12}
Tin (Sn^{++})	1.1×10^{-4}	3.8×10^{-8}
Zinc (Zn^{++})	1.1	2.3×10^{-7}

8.3.1.3 SCALE FORMATION

A significant operating concern in neutralization systems is the formation of scale on process piping, equipment and instrumentation commonly encountered in lime processes (Neven et al, 1985). The composition of scale is similar to the composition of the precipitates formed in the reactions. This is due to either direct precipitation on the vessel surfaces, or entrapment of precipitate in gypsum scale formed due to gypsum supersaturation. Another source of scale formation is absorption of CO_2 from the air by calcium-saturated solutions and precipitation of calcium carbonate at alkaline pH:

$$\mathrm{Ca(OH)_2 + CO_2 \text{-->} CaCO_3 + H_2O} \qquad (8.14)$$

Scale which consists at least in part of carbonates is susceptible to removal by acid treatment, for example by inhibited hydrochloric acid:

$$\mathrm{CaCO_3 + 2\ HCl \text{-->} CaCl_2 + CO_2 + H_2O} \qquad (8.15)$$

Gypsum scale can also be removed by acid, but the removal mechanism is less obvious. Dissolution of entrapped hydroxides and partial conversion to calcium bisulfate may weaken gypsum scale such that it falls or can be knocked off surfaces to which it adheres.

$$\mathrm{2\ CaSO_4.2H_2O + 2\ HOl \text{-->} Ca(HSO_4)_2 + 4\ H_2O + CaCl_2} \qquad (8.16)$$

8.3.1.4 LIME

Two forms of lime are used in industrial treatment processes - quicklime and hydrated lime. Quicklime is the product of the calcination of limestone, and consists primarily of the oxides of calcium and magnesium. Three classes of quicklime are recognized: (National Lime Association, 1976)

1. High Calcium Quicklime - containing less than 5 percent magnesium oxide.
2. Magnesium Quicklime - containing 5 to 35 percent magnesium oxide.
3. Dolomitic Quicklime - containing 35 to 40 percent magnesium oxide.

Quicklime is available in a range of sizes, of which crushed or pebble lime is the most common, ranging from 50 mm to 6 mm.

Hydrated lime is a dry powder obtained by combining quicklime with a stoichiometric quantity of water to form the hydroxide. Its chemical composition reflects that of its parent quicklime. Generally, only a small portion of the magnesium oxide hydrates (5-20 percent). Pressure hydrated dolomitic lime may have almost all of the magnesium oxide hydrated.

Hydrated lime normally is sized in the range of 75 to 95 percent minus 200 mesh.

The lime used in the treatment process is usually added as a slurry of 10-15 % calcium hydroxide in water, termed milk of lime. The slurry may be made by mixing hydrated lime with water, or by slaking of quicklime. Quicklime is cheaper than hydrated lime per unit of neutralizing capacity, and costs less to transport, but it requires the use of slaking equipment, and requires protective equipment for workers since it can cause severe burns. As a rule of thumb, quicklime is used when demand is over 5 tons/day.

When quicklime is used, it must be slaked on site. The slaking reaction is the conversion of calcium oxide to calcium hydroxide through the addition of water:

$$CaO + H_2O \dashrightarrow Ca(OH)_2 \qquad (8.17)$$

This reaction is exothermic, releasing 15.3 kcal/gram mole slaked (National Lime Association, 1976). Because of the heat that may be generated in accidental slaking of lime, bagged quicklime should never be stored adjacent to combustible materials.

Quicklime will deteriorate in storage more rapidly than hydrated lime. In dry storage, hydrated lime in bags may be stored for periods up to one year; quicklime should not be stored over three months. Quicklime will react with atmospheric moisture (equation 8.17) and CO_2 ("air slaking"), while hydrated lime will react with CO_2:

$$Ca(OH)_2 + CO_2 \dashrightarrow CaCO_3 + H_2O \qquad (8.18)$$

8.3.1.5 SLUDGE STABILITY AND DISPOSAL

Sludges resulting from neutralization of ARD can contain gypsum, heavy metal hydroxides, heavy metal arsenates, calcium arsenate, and heavy metal sulfides. Stability is gauged by the potential for various components of the sludge to re-enter the environment. Chemical aspects of the various sludge components in terms of potential for remobilization are discussed in the following paragraphs.

The sludge components are immobile under the conditions at which they are produced and thus the sludge is stable on a short term basis. Instability may occur when the sludge is exposed to air or solutions of differing composition, or when changes occur over an extended time period due to a slow chemical reaction.

Gypsum

Gypsum has a solubility in pure water at 25°C of 2.1 g/L which makes the water "hard" and can contribute a significant dissolved solids loading (Seidell and Linke, 1952). Nevertheless, lime treatment is an accepted method for neutralization of sulphate solutions and normally produces a gypsum-saturated effluent.

Water percolating through the sludge would be capable of dissolving significant quantities of gypsum over time. However, a gypsum sludge, and particularly a mixture of gypsum and metal hydroxides, would exhibit low permeability. Gypsum sludges deposited to minimize exposure to water flow should reduce dissolution to a negligible rate.

Heavy metal hydroxides

The solubility of the metal hydroxides in water increases as pH drops. After long-term exposure to air and groundwater, the sludge pore water pH may drop below the treatment process pH. Thus, potential for resolubilization of the heavy metals exists, and this could dictate the need to treat the sludge as a special waste, requiring special containment to control water passage.

Presence of complexing ions (such as cyanide, chloride, or naturally occurring organic ions) can greatly enhance the solubilities of many heavy metals. Careful selection of the disposal site must be made to ensure that waters bearing such ions do not have opportunity to leach the heavy metals and carry them back into the environment.

The best assurance of stability of these precipitates will come from testwork which exposes the sludge to conditions expected to exist in the disposal area under circumstances that accelerate achievement of equilibrium between the solution and the sludge; i.e., complete suspension in a lab agitator. Analysis of the solutions resulting from such equilibrium contacting will provide an indication of the potential for leaching of the metals. The water used in such a test should be representative of that which will contact the sludge in its proposed storage site.

Arsenic

The question of sludge stability in arsenate control has had a significant impact on treatment process design. Robins and Tozawa (1982) reviewed the stability of calcium arsenate as a product of waste treatment for gold processing waters. Theoretical and experimental results were cited to show that calcium arsenate can break down releasing arsenate ion when in contact with carbon dioxide, bicarbonate, or carbonate:

$$Ca_3(AsO_4)_2 + 3\ CO_3^{=} + 2\ H^+ \text{-->} 3\ CaCO_3 + 2\ HAsO_4^- \qquad (8.19)$$

They concluded that a number of other metal arsenates would exhibit improved stability under atmospheric carbon dioxide, and suggested that their presence might explain why the disposal of mineral processing arsenical wastes is reasonably effective.

Drawing on zinc industry experience, additions of ferric iron have been used to reduce arsenic in solution to low levels while achieving stable sludges. Krause and Ettel (1985) conducted experiments to show that while the simple ferric arsenate, $FeAsO_4$ is unacceptably soluble for the safe disposal of arsenic, "basic ferric arsenates" with Fe/As molar ratios greater than 4 give acceptable solubilities over the pH range of 3-7 and are therefore environmentally safe for disposal. The presence of CO_2 was found not to increase the As solubility. This paper provides the basis for an acceptable arsenic disposal process which hinges on careful neutralization to ensure precipitation of ferric arsenate rather than calcium arsenate, and provision of sufficient ferric iron in the feed to ensure that the molar ratio is high enough to limit solubility.

Generally, in each case, testwork should be completed to ensure that the sludge stability (arsenic solubility) for the process in question is adequate. The chemistry of such sludges is too complex to declare them stable or un-stable based solely on the solubility products of the compounds presumed to exist in the sludge.

Sulfides

If contaminants are precipitated as sulphides, careful consideration must be given to the possibility of acid generation occurring in the deposited sludge.

While the equilibrium solubilities of metal sulfides are low, the precipitates are very fine, have high specific surface areas, and are very amenable to chemical or biological oxidation, with resultant remobilization of metals. However if lime neutralization preceded sulfide precipitation the alkaline characteristics of the combined sludge could counteract acid generation and limit permeability to water and air. Testing of the sludge under conditions that simulate its deposition environment, but favour oxidation would provide a good indication of its long term stability.

8.3.2 Alternative Process Configurations

The treatment process is straightforward; nevertheless, there are alternative methods for accomplishing essentially the same objective. This section discusses the alternative unit operations available and reviews the designs in which they have been employed based on published information.

8.3.2.1 UNIT OPERATIONS

Unit operations alternatives for neutralization and clarification are provided in the following sections.

Stirred Tank Reactors

The stirred tank reactor provides good conditions for neutralization and precipitation by ensuring that the contents are uniform and that "hot spots" of high reagent concentrations are minimised. High local reagent concentrations lead to extreme supersaturation and production of large quantities of fine precipitates that may be difficult te settle. The desired condition is one in which the tank contents are sufficiently close to neutralization equilibrium that the precipitates prefer to deposit on previously precipitated material rather than form new nuclei. This will result in precipitate particle growth, with improved settling properties.

The stirred tank can be constructed in a wide range of sizes, permitting any reasonable residence time to be achieved, according to the dictates of the testwork. More than one tank can be used, either in series or in parallel, to achieve reliability through "installed spare" capacity or to permit staging of the process as described below.

Where physical descaling becomes necessary, the stirred tank provides better access to reactor components than a pipeline reactor.

When desirable, this type of reactor can be air sparged to assist with oxidation of ferrous iron to ferric. The mechanical agitator disperses air throughout the vessel and increases oxygen transfer by maintaining a high gas-liquid surface area.

Over-agitation should be avoided since it results in attrition of the precipitate particles and thereby inhibits liquid-solids separation.

Batch stirred tanks can provide very good process control since neutralization and reagent addition can be conducted in a stepwise manner with pH measurements or analyses conducted between steps or prior to discharge. However, due to the unsteady nature of batch operation and inefficient equipment usage, batch processing would only be realistic for very small operations, where processing occurs infrequently.

Continuous stirred reactors are more common, since they can be operated reliably at steady state with predictable product quality, using minimal instrumentation and operator intervention. Continuous reactors make efficient use of equipment and are easily integrated with a steady feed flow.

By definition the contents of a continuously stirred reactor tank are identical to the product. Therefore it is feasible to operate more than one tank in series to meet various process objectives. For example, a two step neutralization process can improve soluble arsenic removal by maximizing the formation of ferric arsenate. The first reactor can be operated at pH3 which favours ferric arsenate formation over calcium arsenate, while the second tank is operated at pH 7.5-8.0 which ensures efficient heavy metals removal. Staged addition also prevents undesirable high supersaturation which occurs when feed and lime react in a single tank operating at final pH. Reactors in series also reduce short circuiting problems and improve the operation of control systems.

Pipeline Reactors

In cases where the reaction is rapid and/or pumping over an extended distance is required, it may be feasible to inject the reactants at the start of the pipeline and separate the sludge from the treated effluent after discharge at the other end. The pipeline velocity must be sufficient to promote turbulence and keep the reactants and precipitates in suspension. A limited amount of air can be entrained in the line if oxidation of a component of the feed is required.

The pipeline approach may be more appropriate when neutralizing with a soluble reagent such as caustic, since the reaction time will be short and the required pipeline length is reasonable. Short reaction times facilitate more accurate reagent dosage control, since control is usually achieved by a feedback signal from a pH sensor located downstream of the injection point. A caustic neutralization scheme in a pipeline reactor might reduce scaling compared to a lime system, especially if line velocity is low.

Gravity Clarifiers

Although the primary focus of liquid-solid separation in ARD treatment is the clarification of effluent, sludge thickening is also important in terms of reducing the volume for ultimate disposal. Clarification and thickening are sedimentation processes that use similar equipment. Clarification is a relatively inexpensive process for solids separation when large volumes are treated.

Circular thickeners are familiar to the mining industry, and may be used as clarifiers in ARD treatment. Various designs seek to enhance the performance of these basic units, such as plate-type thickeners which multiply the settling area, and reactor clarifiers or hopper clarifiers which optimize flocculant performance. In cases where space is available, clarification can be accomplished in a pond with sufficient capacity to provide clarification, thickening and sludge storage (Coulson et al, 1983).

In the following sections, brief descriptions of some important types of gravity clarifiers will be provided , together with design aspects relevant to ARD treatment. For more information, the references should be consulted directly.

Circular Clarifiers

The basic concept of a continuous gravity thickener is illustrated in Figure 8.3-2 (Metcalf and Eddy, 1972). The feedwell dissipates the kinetic energy in the feed and introduces it to the tank at an appropriate level. The tank provides sufficient residence time and settling area to permit solids separation and sludge densification. The tank normally has a sloped bottom and rake arms to assist movement of the thickened sludge toward a drawoff point. The rake arms also keep the sludge fluid enough to permit movement to the draw-off point and to allow water to escape from the compression zone to the clarification zone. Cone scrapers perform a similar function in the discharge cone which feeds the sludge pumps. The overflow launder collects the clarified effluent for transport to an outlet. The effluent enters the launder over a weir which can be levelled to ensure uniform overflow distribution around the tank periphery.

Design variations reflect the various underflow (sludge) handling schemes that may be selected. Tanks may be:

1) elevated above ground, permitting direct access to the discharge cone and piping, or

2) an access tunnel may be provided for an on-ground tank, or

3) a large centre pump room may be installed within the on-ground tank wherein pumps and piping may be serviced without need for tunnels (King, 1980).

The rake mechanisms may be supported from a bridge constructed across the top of the tank, or (in larger units) from centre columns built within the tank. The drives may be centre driven or peripheral traction driven.

Reactor Clarifiers

ARD treatment plants commonly employ organic flocculants or polyelectrolytes to enhance flocculation and increase settling rates. Flocculation efficiency is improved by adding polyelectrolytes to high solids concentration feed in a "flash mix" chamber followed by a flocculation chamber. Neutralization of ARD feed in a single pass does not produce enough solids for efficient flocculation. Recycle of solids from the clarifier underflow to the feed, upstream of polyelectrolyte addition, is essential to build high density sludge.

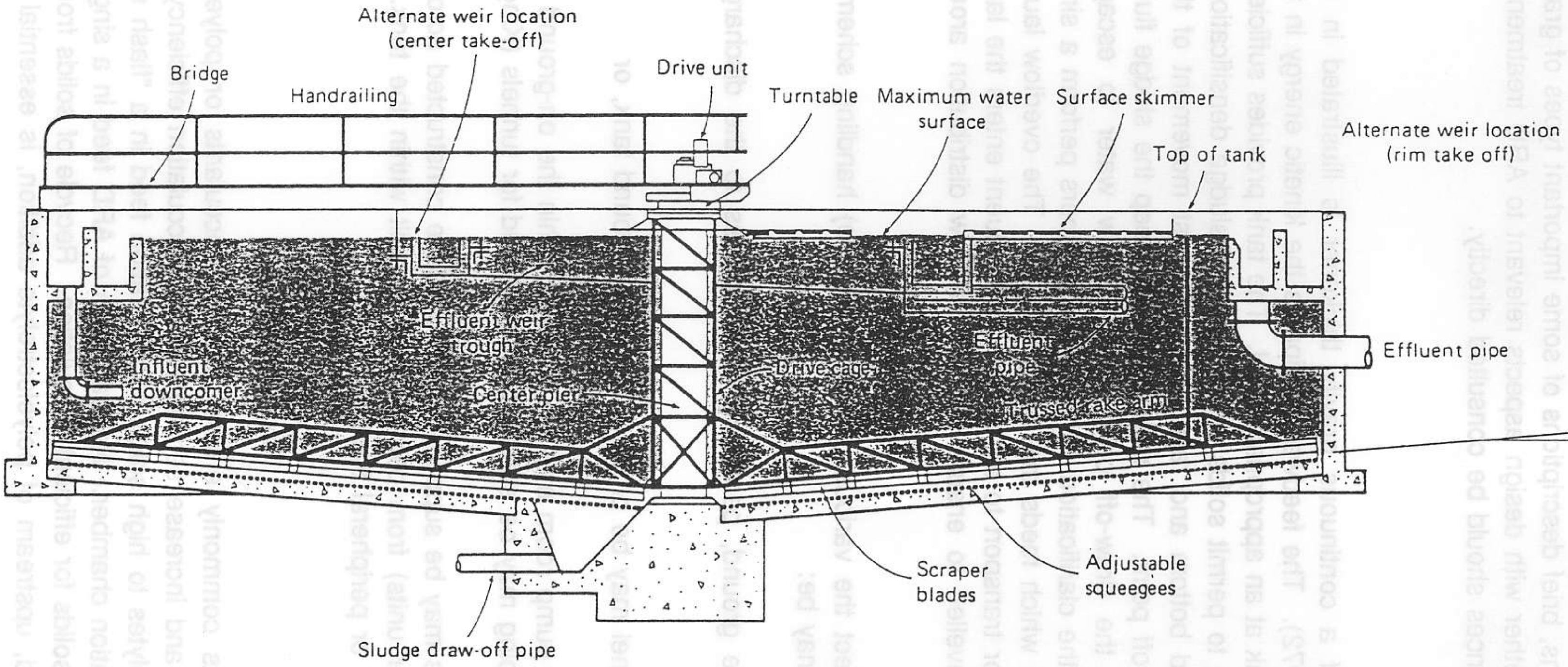


FIGURE 8.3-2
CONTINUOUS GRAVITY CLARIFIER
(after Metcalf & Eddy, 1972)

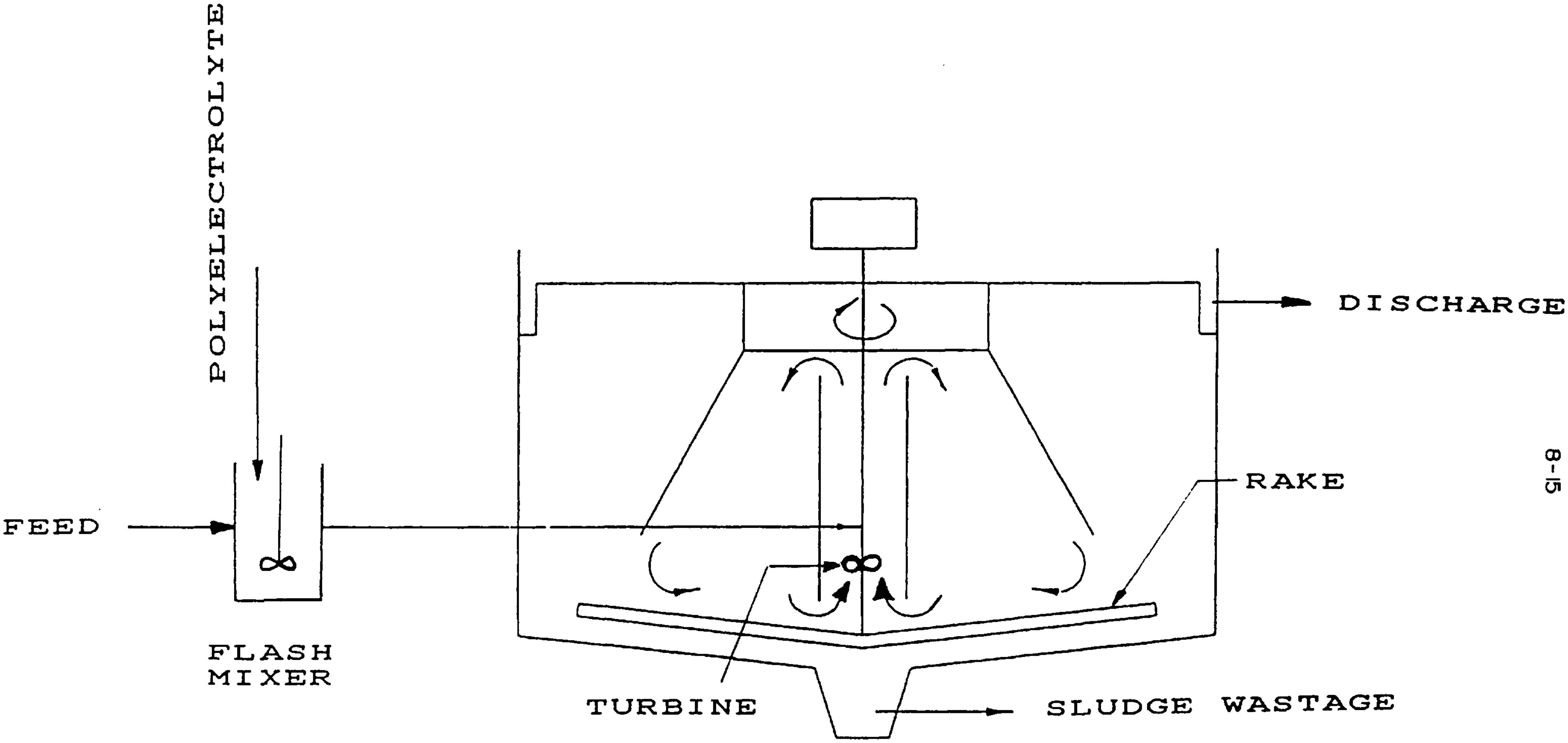


FIGURE 8.3-3
REACTOR CLARIFIER

The reactor-clarifier was developed to provide the flash-mixing, flocculation and sedimentation all in one vessel. A typical unit, shown in Figure 8.3-3, consists of a stacked drive which moves the rake mechanism at a slow speed and rotates a turbine agitator located in the top portion of a centre reaction well at a higher speed. The feed, mixed with flocculant as it enters, is contacted with previously settled sludge in the reaction well by means of the pumping action of the turbine. The high solide concentration which is recirculated provides optimum conditions for flocculation. The mixture is pumped out of the reaction area to the clarification zone where the flocs settle and are raked to the centre to be recirculated once again. As the sludge quantity builds up, small amounts are discharged periodically to maintain the system balance.

The reactor-clarifier may be operated so that the mixture leaving the reaction zone passes through a bed of flocculated solids ("sludge blanket") in the clarification zone which provides a filtering action, and improves clarification efficiency.

Hopper Clarifier

The hopper clarifier was developed by the South African gold industry for preclarification of pregnant cyanide solutions in the late 1970's (Watson, 1988). It seeks to accomplish the same objectives as a reactor-clarifier without recourse to mechanical drives. It is a variant of the sludge blanket clarifier described by Svarovsky (1979). In order to function without a rake, a steep cone angle is required, typically 60°, and this will limit the practical size of the equipment. Because of its simplicity, it is easy and inexpensive to operate, requires very little operator attention, and is virtually maintenance free.

A flocculant and a coagulant are added to the feed, which then enters the clarifier (Figure 8.3-4). The fresh feed enters the turbulent zone and mixes with previously flocculated material, ultimately forming flocs suitable for settling. The turbulent zone is located at the bottom of an inverted cone. As the mixture rises and flares outward, the upward flow velocity decreases and a fluidized floc bed is formed. At the outer edge, upward velocity is negligible, and the flocs settle and move down the sloping side of the cone toward the apex discharge. As with the reactor-clarifier, the floc bed can act as a filter. In processing gold pregnant solutions, the minimum solids concentration in the floc bed for effective filtration is 7%.

Sludge Removal

Sludge removed from the apex is either recirculated back to the feed pipe, or is removed from the circuit. Sludge wastage rate is controlled to maintain a constant sludge bed height (Watson, 1988).

Lamella Clarifier

The lamella principle is based on the idea of multiplication of settling area through stacking of plates (Maki and Smith, 1983). In effect, clarification area is increased by placing a series of clarifiers one on top of the other. By so doing, the floor area required for a clarifier can be drastically reduced. This can be especially significant for use in northern areas where there is strong incentive to house the equipment, and to heat the operating area.

A lamella clarifier with horizontal plates would maximize the clarification area. The plates are inclined at an angle of 45 degrees for high specific gravity solids or 55 degrees for low specific gravity, sticky, or gelatinous materials to direct solids to the thickening zone. Solids are recycled from the clarifier underflow to the neutralization reactor to build high density sludge. High density sludge recycle systems commonly used with Lamella clarifiers have the potential to release supersaturation of gypsum and therefore reduce scaling problems. Lamella clarifiers and similar designs such as tube settlers are prone to poor feed distribution over the plates and to re-entrainment of solids in the feed stream. These difficulties reduce the advantage otherwise expected for the design. Proprietary designs such as the individual throttling orifices and feed ports designed into each plate pack by Axel Johnson help to mitigate the distribution problems.

Lamella thickeners are available either as free standing units or as tank-mounted plate packs. Settled solids dropping off the ends of the plates may be collected by a steep-sloped hopper or raked towards a discharge cone in conventional fashion. The overflow is collected by a series of launders located inside the basin or tank.

8.3.2.2 CASE HISTORIES

Brunswick Mining and Smelting Corp. Ltd.

The waste water treatment plant at Brunswick Mining has been described by Fraser (1988). The plant is designed to handle 1360 l/min (300 IGPM) average and 1820 l/min (400 IGPM) maximum of acid mine water which assays:

pH	2.7	
Pb	2.6	mg/L
Cu	10	mg/L
Zn	1000	mg/L
Fe	1200	mg/L
$SO_4^=$	8600	mg/L

The water is pumped from underground to a 50,000 Imperial gallon surge tank, from which it flows to the treatment plant building where it is neutralized with 18% milk of lime slurry in two 8000 Imperial gallon stirred tanks in series. Aoration is provided to

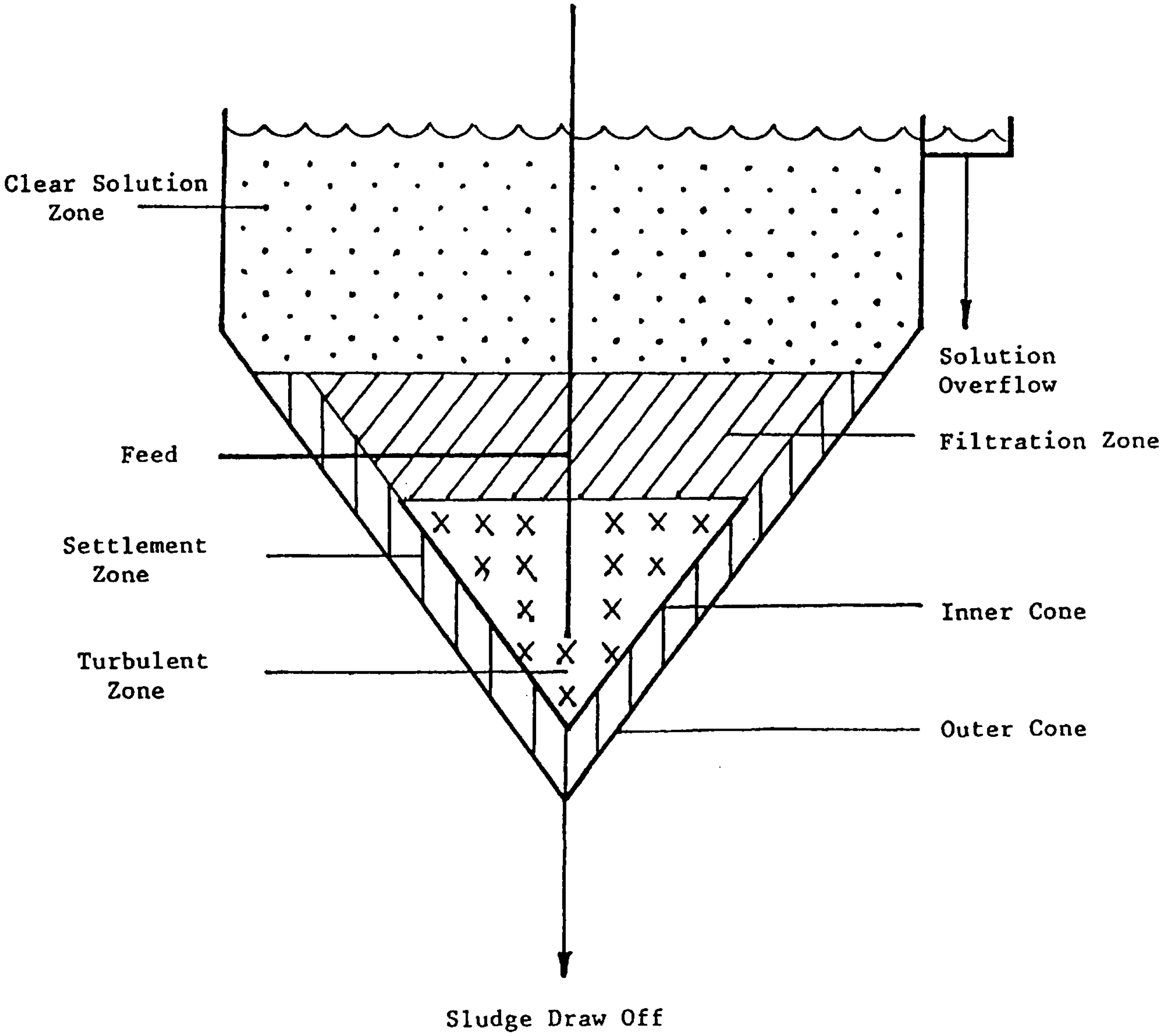


FIGURE 8.3-4
SCHEMATIC REPRESENTATION OF THE
DOUBLE CONE HOPPER CLARIFIER

oxidize any ferrous ions, using 150 cfm per tank. The neutralized slurry flows to a flash mixing cell on the head end of the lamella clarifier, where an anionic polymer is added as a flocculant. The flash mix cell has a retention time of 2 seconds at maximum flow rate. The water then enters the main vessel, a concrete tank in which are suspended two 56-plate packs inclined at 55°. The overflow passes through a turbidity meter and is diverted to a clean water channel. Final pH is 9.5. The underflow thickens to 5-8% solids and is pumped to the mill tailings box for disposal. Approximately 10% of the underflow is returned to the first neutralization tank to act as seed for gypsum deposition. This recycle was instituted primarily as a means of controlling gypsum scale formation in the equipment.

Effluent quality is indicated by the following 1980 yearly averages: (Sampson, 1981)

pH	9.6	
Zn	0.58	mg/L
Fe	0.21	mg/L
Cu	0.056	mg/L
Pb	0.21	mg/L
TWH	2484	mg/L

Les Mines Gallen

This plant is similar to the Brunswick design and size, and benefits from the experience gained at Brunswick (Lecuyer, 1983). In this case, the incoming pH is 3.5, and it is raised to 9.5 by lime addition.

To achieve high density sludge, recycled sludge is premixed with lime before adding to the incoming mine water. Recycle volume can be up to 50% of the mine water flow. This provides a change in the chemistry of the sludge that promotes the growth of larger and faster settling particles. Sludge densities of over 20% solids have been obtained. The sludge is discharged to the sludge basin for storage, and will eventually be covered and revegetated.

The thickener overflow is discharged to a polishing pond, from where it discharges into the Kinojevis river system.

The sludge basin is in a natural depression facilitating construction. The sludge basin and polishing pond required 90,000 cubic yards of excavation.

The thickener tank is a 26 foot diameter by 20 foot high steel vessel containing the lamella plate packs; the reactor tanks are 12 feet in diameter by 16 feet high (Lecuyer, 1983).

Bell Copper

The Bell Mine is an open pit copper mine located on Newman Peninsula, Babine Lake, approximately 65 km northeast of Smithers, B.C. The mine operated a lime treatment plant for 93 days during a required discharge of tailing pond supernatant (Goodwin, 1986). This plant was designed to treat pit water prior to discharge to the tailing pond as a means of reducing one of the largest sources of metal contamination to the pond. Two seepage sources were treated independently in a less controlled manner. Plant operating results are given in the Table 8.3-2

TABLE 8.3-2

BELL COPPER
PIT WATER AND SEEPAGE WATER ANALYSES

			Dissolved Metals mg/L			
Sample Location	pH	Cu	Zn	Mn	Ni	Fe
Pit Water	6.7	0.32	1.2	3.3	0.3	0.1
Lime Treated Pit Water	8.7	0.02	<.1	0.1	0.2	0.1
No. 1 Seepage	6.5	0.72	0.3	2.6	0.1	1.3
Treated No. 1 Seepage	9.0	0.03	<.1	0.4	<.1	<.1
No. 6 Seepage	6.6	2.2	0.4	4.4	0.2	0.3
Treated No. 6 Seepage	9.7	0.03	<.1	<.1	0.1	<.1

The plant was capable of treating 1900 L/min. It consisted of two steel tanks: an upper lime addition tank of 30 m^3 capacity, and a lower mixing tank of 19 m^3, capacity. In the lime addition tank, calcium hydroxide was added to a slipstream from the main flow at the rate of 200 kg per 8 hour shift, and the resulting precipitates were mixed and kept suspended by two agitators. Overflow from this tank gravitated to the mixing tank which carried the main flow through the system. The resulting discharge to the tailing impoundment had a pH of 9-10. The plant was notable in that it depended solely on high pH to reduce dissolved metals to acceptable levels. Alternative processes would usually depend on co-precipitation with ferric iron to reduce metals at lower pH.

Equity Silver

Equity Silver Mines Limited operates an open pit mine and process plant approximately 37 km southeast of Houston, B.C. (Patterson, 1988). Equity collects and treats effluent from the waste and tailings deposits, as well as various structures constructed from waste before the ARD problem was identified. Treatment is accomplished by processing through a lime neutralization plant and the resultant sludge is disposed with mill tailing at a 1:10 ratio during mill operation. Operating results are provided in Table 8.3-3.

TABLE 8.3-3

RESULTS OF WATER TREATMENT, EQUITY SILVER MINE

	pH	Acidity	SO_4	Cu(d)	Zn(d)	Fe(d)
Raw ARD	2.35	10,000	8,500	120	80	800
Treated	7.80	nil	1,600	0.01	0.04	0.03

Pumping Cost: \$0.15/m^3
Treatment Cost: \$0.70/m^3

Approximately 800,000 m^3 of ARD must be collected annually and treated. The treated water complies with the quality objectives and is discharged at dilution ratios governed by metal content and stream flow rates.

ARD is received at the Treatment Plant and is neutralized in two agitated tanks, 3.4 meters diameter by 4.0 meters high. These have recently been converted from parallel operation to series operation. Feed enters the top of one, passes out the bottom into the next, and overflows the second to the sludge pond.

Lime is received as -1/4 inch pebble quicklime, and is pneumatically conveyed into a 200 ton silo. Two 6 inch screw conveyors add lime to two Denver attrition cells set up to act as slakers. Water is added under temperature control to make up the required milk of lime density. The milk of lime is pumped by SRL centrifugal pumps through a settling chamber for grit removal and thence to a storage tank. SRL pumps are used to circulate the milk of lime from the storage tank through a ring main and back to the tank. Valves oontrol the lime addition to the neutralization tanks to maintain the desired pH, normally about 8.5.

Gypsum scale formation on the tanks is a problem, and Equity shut down the system about every 1-1/2 months to remove the scale. This is done manually with air chippers, and requires about 8 hours. If time is available to allow it to dry out, removal is easier.

Neutralized slurry is not processed through a clarifier for solution recovery, but is released into a sludge pond, where the precipitates are allowed to settle out. Submersible sludge pumps transfer the settled sludge to the Tailing Pond. ARD sludge was initially classified as a Special Waste according to the first draft of B.C.'s Special Waste Aot. As a result of recent revisions to the act, ARD sludge appears to be exempt, and consequently, Equity are permitted to dispose of the sludge with the mill tailing at a 1:10 ratio.

Settled sludge reclaim for transport to the tailing pond has met with operating difficulties. If a surface layer of water is present, this is pumped preferentially, with no lateral movement of sludge to fill the cavity initially excavated by the pump. The solution has been to build a divider dyke between the active pond and the pond being excavated. Elimination of the water sources improves the situation, and at 5 percent solids, sludge migrates over the full length of the pond, supplying feed to the pump.

Sullivan

The Sullivan underground lead-zinc mine at Kimberley, B.C. produces from an orebody consisting of galena and marmatite in a siliceous host rock with abundant pyrrhotite (Kuit, 1980). A result of exposing this mineralization to air and water by mining is the production of ARD containing various soluble metals from the oxidized minerals. The activity of the sulphides at Sullivan is so high that most of the soluble iron, the dominant dissolved metal, is present as ferrous ion.

A high density sludge (HDS) process was installed and started up in 1979 to treat effluent from the mine and two tailing impoundments. Its design flow was 18 m^3/min, but 27 m^3/min in spring can be accommodated. Lime slurry and recycled clarifier underflow are blended in a sludge/lime mix tank at the head of the process. The premixing of the lime with the recycled sludge is regarded as of paramount importance to maintaining satisfactory settling properties of the sludge.

The limed sludge passes to a flash mixing tank where it is added to the incoming effluent. It then overflows to a lime reactor where a residence time of 50 minutes allows the reactions to go to completion. The pH in this reactor is used to control the lime addition rate in the sludge/lime mix tank. Air is added in the lime reactor with high shear agitation to convert the precipitated ferrous hydroxide to ferric hydroxide. This results in lower heavy metals concentrations in the effluent, improved settling properties in the clarifier, and best sludge stability.

Flocculant solution is added to the product of the lime reactor and then gently mixed in a flocculator before entry to the clarifier. The concrete clarifier is 36 m diameter, with peripheral overflow. Underflow sludge at 20-30 percent solids is recycled or purged. The recycle ratio is 10-20:1 based on solids precipitated from the feed. Sludge is discharged to an impoundment area underlain by sand and cobble, providing good drainage characteristics without solids escape. No supernatant is discharged from this facility. Plant performance is summarized in Table 8.3-4:

TABLE 8.3-4

SULLIVAN HDS PROCESS PLANT PERFORMANCE

	Feed	Effluent
Fe (mg/L)	250	<1.0
Zn (mg/L)	22	<0.2
Pb (mg/L)	5	<0.1
Cu (mg/L)	0.15	<.05
As (mg/L)	0.02	<.01
TSS (mg/L)	75	<15.0
pH	4.5	9.5

8.3.3 Process Development and Testwork Procedures

Prior to detailed design of a treatment system for ARD it will be essential to conduct process development and treatability test work. Ideally testwork would be conducted using representative waste water samples from an existing operation. This way, testwork results can be directly applied to the design of the required treatment facilities. However, for new mining projects, waste water samples that represent actual operating conditions will not be available. Simulated waste water samples can be generated once ore and waste rock samples are available. Effluents from humidity cells or column tests can be utilized to characterize the waste and to produce simulated samples.

Prior to the availability of samples, one can initiate the conceptual design stage based on known ore and waste rock characteristics and experience at similar operations. It is important to note that conceptual designs often chango dramatically from early planning through to detailed design. The conceptual design is modified and optimized as the project proceeds and better quality information becomes available. Design failures associated with new treatment systems are commonly the result of one of the following factors :

1) Non representative samples
2) Insufficient bench and pilot testing
3) Inaccurate estimation of hydraulic loading
4) Failure to compensate for hydraulic and chemical loading variability 5) Inadequate process control instrumentation.

Obtaining representative samples can be difficult. Good judgement and experience are necessary to ensure that simulated samples selected for the test program are as representative as possible. The following sections outline the characterization and treatability tests required prior to detailed design. Proper care and attention during the testing will prevent some of the common design errors.

8.3.3.1 CHARACTERIZATION OF FEED

As discussed above, representative samples for the testwork program may not be available during the planning phase of the project. Limited data will be provided by the chemical characteristics of the ore and waste rock, the process flowsheet for the mill, and the proposed handling procedures for wastes from the mine and mill. This information should be reviewed prior to commencing the testwork program to ensure that the bench studies are as representative as possible of future operation. Once it has been established from acid-base accounting that the waste rock and/or tailings has a potential to generate acid, planning for the necessary bench tests should proceed with a review of the following items.

8.3.3.1.1 Characteristics of Ore and Waste Rock

The ore and waste rock can be characterized by examining the following information obtained from samples; namely:

1) physical characteristics, (hardness, pore water pH, etc.)
2) mineralogical characteristics (e.g., pyrite, pyrrhotite, calcite, arsenopyrite, galena etc.),
3) metal composition, (iron, copper, zinc, arsenic, mercury, etc), and
4) composition of major anions (sulphate, phosphate, carbonate, chloride, arsenate, etc.).

Examining analytical data from ore and waste rock samples will establish a sound basis for planning the testwork program and will assist in determining the need for additional unit processes in the treatment system such as oxidation, reduction, cyanide oxidation or sulphide precipitation. Neutralization by itself may not be sufficient to achieve acceptable effluent criteria. Metals that may require treatment processes in addition to neutralization are chromium, mercury, arsenic, antimony, iron, and manganese. Information on the variability of the ore and waste rock would also assist in characterizing the potential feed to the ARD treatment plant.

8.3.3.1.2 Mill Flowsheet

In addition to examining ore and waste rock characteristics it will be essential to review the proposed metallurgical extraction processes in the mill; i.e. the use of cyanide or flotation chemicals, and the use of chemical processes such as pressure oxidation, and acid or bio-leaching. For example, pressure oxidation of sulphide ores will solubilize metals such as mercury and arsenic, while the use of cyanide will result in the complexation of metals such as copper and nickel. Treatment of mill effluents containing cyanide would be essential to comply with existing regulations for discharge, and cyanide would not normally be included as a major contaminant in ARD type waste streams. However, low levels of cyanide can persist in treated mill effluents and ultimately report with ARD wastes from tailings impoundments. In addition low levels of complexed metals (mainly copper and iron) may also accompany ARD streams originating from tailing ponds containing cyanide.

8.3.3.1.3 Handling Procedures for Waste Rock And Tailings

Information on the characteristics of the feed to the ARD treatment plant will be provided from plans for handling waste rock and tailings. Important points to consider will be:

1) Open pit versus underground operation. Open pit operation will likely encompass larger volumes of waste rock that must be stored outside the active pit while the quantity of waste rock removed from an underground mine could be small.

2) Method of tailing storage. A flooded tailing pond would tend to reduce acid generation, while a partially flooded pond could produce an appreciable quantity of acidic drainage due to exposure to air.

3) Groundwater, surface water, and seepage characteristics. Ideally the movement of groundwater through tailing ponds or waste rock piles should be minimized. Hydrological studies should give some indication of the quantity of drainage which might need treating.

8.3.3.1.4 Production of ARD Feed Samples

ARD treatment plant feed can be generated using waste rock samples once representative solids samples are available. Effluent from column acid generation tests can be used to produce ARD waste water samples from individual waste rock and tailings samples. Typically during the leaching test with a column, the acid production rate will increase with time and reach a peak followed by a gradual reduction. Testwork should be conducted using worst case conditions to demonstrate the capability of the system to meet discharge criteria. A worst case condition sample would be produced by operating a column with alternating wet and dry cycles and collecting the first portion of sample to pass through the column after the dry cycle. Under field conditions ARD characteristics may exceed worst case samples generated in the lab, therefore good judgement and experience should be utilized in determining what the actual worst case conditions may be. Design of the plant will reflect a large number of factors in addition to the lab or pilot test results. Regulatory approval will be based on compliance under worst case conditions. Characterization of the feed will include a number tests as described below.

8.3.3.1.5 Analytical Results

A full set of analyses should be performed on the feed sample, prior to proceeding with the process development bench tests. Parameters would include:

1) major anions, such as phosphate, sulfate, chloride, nitrate, etc.,
2) physical parameters such as acidity, pH, conductivity, and
3) an ICP scan of total and dissolved metals.

8.3.3.1.6 Neutralization Curve

The neutralization curve is generated to estimate quantities of alkali required to neutralize the ARD stream and to provide a basis for conducting batch neutralization tests. The acid content of a waste stream is moasured by titrating a sample of waste against a standard alkali to a specified pH end point. Standard practice is to generate a plot of pH versus quantity of alkali added. This plot commonly referred to as a neutralization curve, provides information on the quantity of alkali required to achieve a desired given pH. The plot can be generated using a soluble alkali such as sodium hydroxide and then converted to other sources of alkalinity such as hydrated lime or limestone.

8.3.3.1.7 Neutralization Reaction Rate

Once the quantity of lime or other alkali has been determine it is important to determine the speed of the neutralization reaction. This test can be accomplished by simply adding the estimated quantity of chemical i.e. lime to a batch mixed sample and monitoring the change in pH with time. The length of time required to approach an equilibrium pH is directly related to the retention time required in the reactor system.

8.3.3.2 SMALL SCALE BATCH PRECIPITATION AND COAGULATION TEST

An initial assessment of the neutralization process can be made by completing a small scale batch neutralization to pH 8.5 on a feed sample. The resulting precipitate is flocculated with polyelectrolyte at 1 ppm concentration. This test should use the actual reagents for the proposed treatment system. The results of this test are used to establish the test conditions for more detailed bench tests. If elevated levels of certain parameters remain in the supernatant after this test it is important to establish whether the contaminant is in suspension or solution.

If a problem is believed due to turbidity, it will be necessary to optimize coagulation and flocculation as described below.

8.3.3.2.1 Flocculation Test and Polyelectrolyte Screening

Required reagent dosages including polyelectrolyte can be determined by conducting screening tests. These tests are conducted using a gang mixer commonly referred to a "jar" tester. The equipment consists of variable speed multiple paddle mixers ganged together in one apparatus. It is commonly used for water treatment process development, and allows a number of reagent types and dosages to be tested at the same time. Side by side comparisons of settling rate, supernatant clarity and sludge compaction characteristics can be made. At the process development stage this may not be necessary, if standard procedures and reagents yield satisfactory results. Further optimization should be left until plant start-up using actual samples. The next step in the process development would be to conduct settling tests to verify clarifier design overflow rate.

8.3.3.3 SETTLING TESTS

Settling test are conducted to verify the following:

1) the design overflow rate for clarification
2) supernatant quality
3) sludge compaction characteristics

8.3.3.3.1 Standard Settling Rate Procedure

A settling test is conducted by filling a graduated cylinder with flocculated slurry and timing the fall of the sludge-solution interface. A standard procedure for conducting such a test is given by Rogers and Poling (1978).

In some situations the clarity of the supernatant will be unacceptable and it will be necessary to evaluate the benefits of sludge recycle. This can be accomplished by incorporating the sludge from a series of settling tests into subsequent second , third and fourth passes. The volume of sample used in each step is reduced as the quantity of sludge increases. The bench test cannot fully evaluate a high sludge density recycle system since the column tests cannot simulate depth provided in a clarifier and the resultant sludge compaction that takes place. However the bench test will indicate whether recycle will enhance settling and clarification. Industrial experience should be relied on for final sludge density prediction and effluent quality from a high sludge density recycle system.

Further "jar tests" may be necessary to investigate other flocculants, optimize reagent dosages, and enhance effluent quality if the results obtained from the settling tests are unsatisfactory.

8.3.3.3.2 Translation of settling test data to design criteria.

Test data that indicates overflow rates above standard industrial practice are not necessarily utilized in design. Equipment vendors are reluctant to support applications where overflow rates are above conventional values for similar applications. If a performance guarantee is required for a bid on the supply of clarification equipmant the vendor will utilize conservative values for overflow rate. Settling tests are conducted to confirm that the process has the capability of meeting discharge criteria. Optimization takes place after the equipment is installed and in some cases a well designed and operated high sludge density recycle system will have a total capacity greater than the design value. However, industrial experience has indicated that treatment systems are commonly over-loaded and the use of conservative design parameters is adviseable.

8.3.3.4 SLUDGE CHARACTERIZATION AND STABILITY

Physical characteristics

The physical appearance of sludge generated by the column settling test in terms of granularity or presence of amphorous material can provide a reasonable indication of the expected properties of the sludge for a single pass system. However, with a high density sludge system it is difficult to duplicate the sludge in the lab using batch tests since the sludge produced by the high density system will result from a high degree of recycle and compaction.

The physical characteristics of sludge from a recycle system are best obtained from experience with similar operations. However, the benefits of recycle can be demonstrated from the settling recycle simulation test described above.

Leaching characteristics

The chemical stability of the sludge can be evaluated using leaching tests conducted at various pH and retention times. If the sludge is stable, the soluble liquid phase associated with the sludge should be the same chemically as the soluble phase of the supernatant quality assuming the pH is the same.

An example of a sludge leaching test is provided by the BC Ministry of Environment Special Wastes Extraction Procedure.(B C Ministry of Envirenment and Parks, 1988).

Leaching tests should be conducted in conjunction with an evaluation of the physical and chemical characteristics of the disposal site.

Factors to investigate would include:

Location of water table
Groundwater movement
Surface water movement
Exposure to air
Erosion potential
Chemical characteristics of the soil including pH, acidity and organic content.

A report prepared by Wasserlauf (1987) on sludge characteristics from Canadian mine/smelters indicated good long term stability at neutral pH . It was noted that sludges from the lime precipitation process generally carry excess lime that will maintain alkaline conditions over the long term.

Sludges containing arsenic should receive extra attention due to potential release of soluble arsenic from calcium arsenate under alkaline conditions. Sludges containing sulfides should also be evaluated to determine the potential for acid generation. Sludges containing elevated mercury would also be a special situation due the potential release of mercury as a methyl mercury or from hydrolysis of mercury hydroxides. Speciation of the mercury would assist in evaluating long term stability in this case.

8.3.3.5 BATCH OXIDATION TESTING

Batch oxidation tests would be conducted when there is a need to oxidize compounds such as ferrous iron or arsenite in order to achieve satisfactory removal. Oxidation using air is sufficient for ferrous iron oxidation at neutral pH or higher : however the presence of ferric iron or other oxidizing agents such as hydrogen peroxide or SO_2-air^x may be required to oxidize arsenite if ferric iron is absent. The oxidation test should be conducted on a simulated waste water sample that has been neutralized to pH 8-8.5 with lime. (The oxidation of ferrous to ferric is slow under acidic conditions due to the solubility of ferric iron.) The batch oxidation test would proceed as follows:

1) Select a sample of simulated waste water that represents worst case conditions. Measure and record sample volume.

2) Neutralize the sample to pH 8.0 with lime, allowing sufficient time for the pH reaction to go to completion.

3) Aerate the sample using air supplied by an air pump, provide sufficient air to maintain a measurable dissolved oxygen residual.

4) Measure and record pH and dissolved oxygen values during the test.

5) Collect samples aliquots with time and

6) Analyze the samples collected for the desired parameter, i.e. ferrous iron, arsenite etc.

7) Maintain pH above 7 during the test via lime addition and record consumption.

8) Plot the concentration of the desired parameter with time to determine the reaction rate of the oxidation step.

9) Test the product using the settling test as described above.

Design of equipment to accomplish oxidation will be controlled by:

1) the oxidation reaction rate and
2) the oxygen transfer rate.

The testing should identify a desired dissolved oxygen concentration that will permit the reaction to go to completion in a reasonable residence time. The reactor system is then sized to provide the required residence time plus a suitable factor.

The oxygen demand of the feed is known from its composition. From this, and the residence time, a desired oxygen transfer rate is calculated. This transfer rate may be used to select the required aeration rate based on an oxygen utilization efficiency determined by experience or consultation with agitator vendors. The agitator is then sized to disperse this much air and transfer oxygen at the required rate based on experience with similar slurries and/or advice from vendors.

8.3.3.6 TESTING FOR OTHER UNIT OPERATIONS

Mercury

Sulphide addition for mercury precipitation would be conducted on a batch basis using a jar tester and varying dosage and retention time.

Arsenic

Conventional practice for arsenic removal is to use ferric sulphate precipitation followed by flocculation and solids removal in a high sludge density recycle system. This is similar to conventional ARD treatment. Ferric sulphate addition for arsenic removal would probably not be required for an ARD waste since this stream should contain adequate iron to precipitate arsenic. Additional treatability tests for arsenic would only be conducted after conventional neutralization and precipitation failed to yield an acceptable final effluent quality.

Problems associated with arsenic removal from an ARD treatment plant would probably be due to either inadequate solids removal or improper neutralization and not a lack of iron. It may also be important to investigate the oxidation state of the arsenic present if sludge recycle and/or step neutralization fail to yield low arsenic residuals.

The iron to arsenic weight ratio should be above 5:1 to achieve good removal. A high sludge density recycle system would benefit solids removal and thus enhance arsenic removal, while stagewise neutralization could benefit precipitation of arsenic, by producing a stable ferric arsenate precipitate as described previously.

A two step neutralization process can improve soluble arsenic removal by maximizing the formation of ferric arsenate. In the first step the acid stream is neutralized to pH 3, to favour ferric arsenate formation.(Ferric arsenate has a minimum solubility at pH 2). In the second step, neutralization to pH 7.5-8 takes place which favours precipitation of the remaining ferric iron as ferric hydroxide and co-precipitation of the remaining ferric arsenate. Sludge produced by this method should be stable in that the quantity of calcium arsenate produced is low.

8.3.4 Design

Design of an ARD treatment plant will be based on a combination of design criteria and standard engineering practice.

8.3.4.1 DESIGN CRITERIA

Design criteria include a large number of items such as pH, feed rate and clarifier loading. The criteria selected for design purposes consider both bench tests results and industrial experience. Items include:

Feed Rate and Hydraulic Loading

The size of the ARD treatment plant will be set by the maximum design hydraulic loading. This design value will be derived from a site specific hydrology and water balance study. Depending on the specific site conditions, the rate of production of ARD will vary considerably. The ARD treatment plant should be preceded by an equalization pond with sufficient size to smooth out short term peaks and ensure that the plant operates at a constant flowrate.

pH

The optimum pH will be based on bench test results but probably will fall in the range of 7.5 to 8.5 depending on the heavy metals present in the ARD feed.

Residence Time

The reactors are sized to provide the required residence time, typically in the range 30 to 60 minutes although shorter residence times may be used if compensation is provided by high agitation and multiple tanks. Two and preferably three tanks are connected in series to defeat short circuiting.

Reagent Dosages

Lime dosage will vary according to the acidity loading on the treatment plant. The variation in loading will be dependent on a large number of seasonal factors. The reagent dosage criteria selected for design will focus on the maximum and minimum dosage values.

Polyelectrolyte dosage will be in the range of 0.5 to 2 ppm depending on the application and the chemical selected. Operators commonly re-evaluate polyelectrolyte dosage and type to maximize performance.

Clarifier Loading

Selection of a design hydraulic loading is based on a combination of settling test data, practices and experience at existing operations, and recommendations provided by equipment vendors. Design criteria for clarification equipment is generally conservative due to industrial experience in handling hydroxide type precipitates.

The overflow rate obtained in the bench test might be as high as 1.5 USgpm/ft^2 under ideal conditions. However equipment suppliers generally will not support equipment design based on high rates since experience has indicated that clarifiers operated at high rates are prone to upset if any of the process or operating conditions change. The rule of thumb for design of clarification equipment is for overflow rates in the 0.5 USgpm/ft^2 range. Conservative design values reduce the probability of failure.

Aeration and Agitation

Design criteria for reactors in which air is to be dispersed for oxidation is a very complex subject, involving estimation of required dissolved oxygen level, oxidation rates, and mass transfer coefficients. This type of analysis is rarely undertaken in practice. Instead, a chemical demand for oxygen is calculated based on the reactions desired, and the rate at which they must proceed to be complete in the desired residence time. This rate is used with a rule of thumb oxygen utilisation from dispersed air (typically 10-20 percent) to estimate the rate at which air must be supplied.

Agitator vendors are asked to recommend an agitator that will disperse the required amount of air in the reactor, comfortably below the flooding point.

Agitation criteria without the requirement for aeration is straightforward and is generally based on tank volume and fluid viscosity which is similar to water until percent solids increase above approximately 5%.

Recirculation Rates

The recirculation rate for sludge is controlled by sludge pumping capacity. Separate sludge wastage pumps are advisable as a means of ensuring constant sludge recirculation rates. A typical sludge recycle would be 5-20% of the feed flowrate.

8.3.4.2 DESIGN PROCEDURES

8.3.4.2.1 General

The design sequence consists of the following steps:

- Review testwork and relevant industrial experience
- Outline the process concept by deciding items such as high density sludge versus single pass, lime versus limestone, etc.
- Lay out the process flowsheet
- Balance flows and solids loadings, calculate reagent addition rates and check process chemistry on the flowsheet.
- Calculate equipment size based on the process flows and design criteria established in testing
- Compile a list of the major equipment
- Complete equipment layout drawings that define the major structural and mechanical requirements
- Complete detailed engineering as required to specify and procure the equipment, select contractors and build the plant.

The engineering will not necessarily proceed in as defined a sequence as laid out above; in reality, there is usually some iteration, as layout constraints may alter the flowsheet, etc. However, the lime treatment process is fairly straightforward, so that few complications would be expected. The following sections discuss some of the pertinent considerations when setting the process concept and selecting the major equipment.

8.3.4.2.2 Reagent Addition

Reagent addition will include equipment for preparation, storage and metering. Individual reagents and associated equipment are discussed in the following sections.

Lime slaking

Burned quicklime is usually received in bulk truckloads as minus 6 mm material. It is pneumatically conveyed to a silo, which should be equipped with a baghouse to remove the lime dust from the transporting air.

Lime slaking is usually accomplished in one of three ways : detention slakers, paste slakers, or lime mills. Paste slakers are said to provide a higher yield of slaked lime owing to the higher temperature at which they operate. Lime mills offer the advantage that grit removal is not required, since it is ground up and discharged with the product slurry. Lime slaking alternatives plus storage, pumping, and materials of construction are discussed in National Lime Association Bulletin 213 (National Lime Associations, 1976).

Limestone

Limestone may be considered to perform a part of the neutralization for larger systems. However, it means providing and operating two systems instead of one, since the desired pH cannot usually be attained in a reasonable time with limestone alone.

Limestone should be added as a ground slurry, to ensure best addition control and optimum reactivity. It may be purchased pulverised, or it may be necessary to reduce its size prior to feed to the grinding system. Limestone is not particularly hard, and is not corrosive, so that the milling circuit should present no particular problems for engineers and operators from the mining industry.

Flocculants

Flocculants used in lime treatment plants usually will be of the polyacrylamide type, although some natural products are used as well. They are quite soluble in water, but their long chain polymeric nature makes the solid particles tend to stick together if initially wet in a mass, so that so-called "fisheyes" are formed in the stock solution. These may take a considerable time to completely dissolve. Therefore, proprietary mixing equipment is sold by the chemical suppliers and others, which is designed to individually wet each flocculant particle before discharging it into the mixing vessel. These range from simple funnels on educators connected to the feed water flow to complex completely automated pneumatic feeders with wetted wall columns. As an alternative, flocculants can be purchased as premixed fluid dispersions which combine readily with water.

The long chain nature of the polymers makes the solutions very viscous, and thus stock solution strengths are limited to about 0.2 to 0.5 percent. At this strength, the solution may be stored for a day or so; eventually it degrades due to polymer hydrolysis. Before addition to the process, the solution is diluted to 0.05 percent, which allows the polymer to fully "unwrap" and hydrate for maximum effectiveness. Ideally, 30 minutes residence time should be provided for hydration after dilution prior to adding the solution to the process.

In order to minimize polymer chain breakage, shearing of flocculant solutions is minimized at all stages. This is a consideration in agitator selection as well as pumping; typically, progressing cavity pumps are used.

Flocculants are not corrosive, and special materials selection for the handling equipment is not required.

Coagulants

Coagulants are usually strong acid salts of iron or aluminum, and thus have an acid reaction in solution. Accordingly, special materials must be selected for the handling equipment. For sulphates, rubber lining, plastic lining, FRP, or stainless steel 316 could be considered. For chlorides, stainless steel would be a poor choice, but the others cited could be used satisfactorily. Chemical suppliers provide brochures recommending handling methods, and an extensive discussion is provided by Lanouette (1977).

In some cases, these reagents are waste products from other industries, and may contain significant quantities of insoluble solids. The need to be able to deal with suspended solids must be borne in mind by the engineer in the design of the reagent mixing and distribution system.

Miscellaneous Reagents

Caustic soda

Caustic soda is usually delivered as a 50 percent solution in tank trucks or drums. Alternatively, It is available in flake form, but at a significant price premium. In remote areas, the cost of transportation may dictate the use of flake.

Caustic soda solution as delivered has a crystallisation point of 12°C, and thus must be stored indoors or in a heated tank at the majority of B.C. locations.

When flake is dissolved, a considerable amount of heat is evolved. This should be recognized in terms of equipment design and safety considerations. Caustic embrittlement of steel is a factor in the design of handling systems when higher temperatures may be encountered, and may dictate more expensive materials of construction.

Soda ash

Soda ash is available in bulk trucks or in bags. Unloading of bulk trucks to a silo is done pneumatically as for quicklime. Soda ash dissolves readily and usually may be handled without any special materials of construction considerations.

Sodium sulfhydrate

Sodium sulphide (Na_2S) and sodium sulfhydrate (NaHS) can be used interchangeably as sources of sulphide ions for water treatment. Because of their molecular weight relationship, sodium sulfhydrate is the more economical. It is available in flake form in drums as well as 45 and 70 percent solution in bulk shipments. The 45 percent solution freezes at 13°C, and so must be treated like caustic solution for storage. Chemical suppliers should be contacted for information on unloading, handling and storage. This material is a potential safety hazard due to evolution of hydrogen sulphide when contacted with acid.

Sulphuric acid

Sulphuric acid is usually sold as 93 percent acid, in drums or in bulk tankers. Use in an ARD treatment plant would likely be in small quantities, for pH adjustment downwards after treatment and prior to discharge. Thus, drum supply might be preferred.

Sulphuric acid can be piped in standard steel pipe and fittings, with due consideration for flow velocities. Excessive velocity will remove the protective ferrous sulphate coating from piping. Other considerations could be use of heavier pipe than usual and back-welded fittings, stainless steel valve trim and pumps, and special design of pump bases to deal with leakage. A review of standard techniques for acid distribution is recommended prior to commencing a design with this potentially hazardous chemical. Contact suppliers for information.

Descale reagents

Inhibited hydrochloric acid may be used for descaling equipment where physical descaling is impractical. Hydrochloric acid normally would attack steel equipment rapidly, but proprietary compounds are available for addition to the acid to limit this corrosion. Contact acid suppliers for more information. Sulphamic acid has been proposed for descaling; it is available in solid form in bags.

8.3.4.2.3 Reactor Design

The reactors are sized to provide the required residence time. Internal feed and/or exit baffles can be added to help eliminate short circuiting but can cause problems for descaling. Best practice will keep the reactor internals as simple as possible.

Solids suspension is not usually a major consideration in these designs, as the solids are typically very fine and suspend easily. Agitator vendors will provide recommendations on selection of agitators and the best arrangement for sparging air under the impeller if aeration is required.

Once the feed is neutralized, its corrosive properties diminish. Depending on the pH at which the first stage is operated, it may be possible to construct all the tanks in mild steel. Alternatively, it may be desirable to make the first stage of stainless steel. Polymer linings or free-standing FRP should not be considered, as they will be damaged by periodic descaling operations.

Concrete tanks are also used. These may be square, rectangular, or circular in construction. Short circuiting can be reduced in a rectangular tank by its division into several compartments, each with its own agitator.

8.3.4.2.4 Clarifier Design

Selection of the type of clarifier will be the result of the experience and preference of the designer, the recent experience of others with similar plants, and an economic evaluation based on solicitation of quotes from vendors. In the high density sludge process, flocculation is critical to successful performance, and thus, separate reactors may be provided for "flash" mixing of flocculant with the feed followed by gentle agitation while the flocs form and consolidate. Alternatively, flash mixing and flocculant may be provided by addition of the flocculant to a pipeline or launder upstream of the clarifier. Sometimes some experimentation with the addition point and method is required after start up, since the hydraulic conditions in a pipeline or launder can be hard to define. Recommendations for design of flash mixing and flocculation facilities are provided in an EPA Process Design Manual (U.S. Environmental Protection Agency, 1975).

8.3.4.2.5 Sludge Thickening, Recycle and Disposal

Sludge thickening is normally conducted in conjunction with clarification. Clarifiers can be provided with a separate thickener zone if a reduction in sludge volume is necessary. Conventional clarifiers normally provide sufficient thickening for the HDS process. Beyond this, the degree of thickening required will be directly related to the disposal requirements. If sludge is to be permanently stored in a single large pond, then separate thickening will not be necessary since the sludge will compact to at least 15 % solids with time. At the other extreme, if sludge is to be trucked off site, then thickening and dewatering using pressure filters may be necessary to control trucking costs.

In terms of design, equipment handling sludge such as rakes and sludge pumps should be selected with these ranges in mind. Industrial experience with similar equipment should be referred to for recycle systems. An adequate flow range for the sludge pumps is essential, since the sludge density can vary widely.

8.3.5 Process Control and Operation

8.3.5.1 REAGENT DOSING SYSTEM AND CONTROL

Lime

Metering equipment should be selected to provide both the quantity of reagent needed at maximum loading and a reasonable degree of control under low feeding rates. Metering equipment with narrow operating ranges and poor controllability at low feeding rates should be avoided. Back-up and standby reagent metering equipment is advisable. Maintenance aspects and industrial experience with similar equipment should be the primary considerations in selecting metering equipment. The reagent dosing system should include flow indicators and alarms in addition to a dosing control system based on an appropriate process variable.

Lime is added as a slurry, which makes continuous controlled addition at low rates difficult. To maintain adequate velocities, periodic additions at full flow are often used. The controller may set the frequency of pulse or pulse length or both. Automatic valves should be selected suitable for slurry service. Branch lines are short and preferably come vertically off the ring main so that solids settle back into the main when flow is shut off. The valve is then mounted in a short horizontal run so that a column of solids cannot settle on the control element, preventing its functioning.

Control of pH and minimization of supersaturation which can lead to scaling will be enhanced by staged addition of lime in a series of stirred tanks. Control also requires sufficient tank residence time to give stable operation. A minimum of 20 minutes per tank is suggested.

Flocculant Addition

Flocculant addition is not usually automatically controlled. The diluted flocculant solution is metered through a needle valve, and its addition rate is monitored with a rotameter or by setting a metering pump. For smaller systems, stock flocculant solution can be diluted in batches; in larger systems, it will be more convenient to dilute continuously. Flocculant performance will be enhanced if dilution is accomplished with sufficient residence time (30 minutes) to allow complete hydration of the polymer before addition to the process.

8.3.5.2 OXIDATION CONTROL

Control and monitoring of an oxidation process can be based on redox potential or dissolved oxygen (assuming air oxidation). Since ferrous/ferric oxidation usually predominates a low redox potential would indicate the presence of ferrous iron. The redox potential should increase with the degree of oxidation, however it may not reach a high potential under neutral pH conditions due to precipitation of ferric hydroxide. Redox potential control would also apply in situations where oxidizing agents such as hydrogen peroxide or SO^2-air are used, since a positive residual of the oxidizing agent should result in a readable positive redox potential. Dissolved oxygen monitoring and control would be feasible in an air oxidation system, since a positive dissolved oxygen residual would indicate that the immediate oxygen demand from ferrous or sulphide oxidation had been satisfied.

In most systems the feasibility of using either redox or dissolved oxygen in a control system is questionable, since the cost savings may not be significant. However the use of either redox or dissolved oxygen instrumentation coupled with a monitoring and alarm system could assist the operator in precess control and failure identification. Two or three tanks connected in series are preferred in order to reduce short circuiting.

8.3.5.3 TURBIDITY MEASUREMENT

Process upsets from ARD and similar types of treatment plants are commonly associated with an increase in the level of suspended solids in the effluent. The condition can be monitored using an on-line turbidity meter coupled to an alarm system. Sampling and analyses of effluent samples during start-up can provide a correlation between suspended solids, turbidity and a number of other parameters. In this way a reasonable instantaneous measurement of general effluent quality can be provided and serious upset conditions avoided. Some maintenance and re-calibration of the turbidity meter is essential in order to provide good quality data.

The use of an on-line instrument such as a turbidity meter can reduce the amount of sampling and analytical work required for process control, and for regulatory compliance. Automatic control can be used to divert the product stream back to the feed source when turbidity specifications are exceeded.

8.3.5.4 ANALYTICAL SUPPORT

At most mine sites, analytical capability will probably be limited to atomic absorption spectrophotometry (AA), and simple titrations. Plant operators can be trained to complete this type of analysis, but the levels of detection achievable often are not adequate for monitoring treated solutions. Clean facilities are required to avoid contamination of assays, and independently determined results can be more credible for external reporting.

Thus, normal analytical support consists of a mixture of on-site determinations for day to day plant control, together with external assays for reporting purposes. Commercial laboratories have equipment such as inductively coupled plasma spectrophotometers (ICP) that may be capable of lower detection limits than AA and can perform multi-element assays more quickly and cheaply. These labs also have personnel trained in the more difficult wet techniques required for some circumstances.

In a larger facility, on-line analysis may be used for process control and monitoring. Where this is the case, the equipment still requires calibration, and this may involve check samples sent to outside facilities. Similar checks should be run on in-house analyses for quality control.

The frequency of sampling will be specified in the permit under which the facility operates. Beyond this, when a treatment plant is in stable operation, samples once a shift are the most that might be required, and this might be reduced to daily sampling or less if feed flow and chemical makeup do not vary quickly. This could be the case where large storage ponds precede the plant.

8.3.5.5 OPERATOR DUTIES

At some sites, there may be no permanently assigned operator for the treatment plant. Operation will be undertaken on a part time basis by an individual whose primary responsibility is elsewhere.

Tasks that must be completed in the operation of the treatment plant could include:

- maintain instrumentation (clean pH probes, calibrate, etc.)
- descale equipment, clean lamella plates
- mixing of reagents
- catch appropriate samples, label and preserve
- arrange for receipt, storage of incoming reagents
- visual monitoring of process (slurry colour, density, etc.)
- control recycle rate for optimum process performance
- dispose of lime grit
- mechanical maintenance, lubrication

The process operator should be encouraged to take responsibility for operation and report preparation and to provide monthly status reports to management on plant performance and potential areas of improvement; i.e., equipment modification, maintenance requirements etc. External reports would be written by a more senior individual.

8.3.5.6 MAINTENANCE AND REPAIR

In a larger facility, maintenance would be completed by qualified tradesmen, either available elsewhere on site, or brought in from outside as required. Functions which could require maintenance beyond that performed by an operator are:

- pH, redox, turbidity instrumentation calibration and repair
- agitator, rake mechanism, blower alignment
- pump repacking/replace mechanical seal
- most electrical repairs
- motor control, motor changes

8.3.5.7 TROUBLESHOOTING

A sampling program should periodically characterize the feed from a process operation standpoint. Any changes in quantities or relative ratios of contaminants should be flagged and analyzed as to their likely impact on future treatment performance. These samples should be taken close to the source to provide early warning, giving time to decide on and implement any changes in operating strategy that might be dictated.

Bench scale process simulation when problems arise will enable comparison of the results with those in the plant and determine if the anomaly is due to changes in feed quality or to equipment malfunction. If these simulations are run on a regular basis, there will be a database of experience built up which may prove useful in troubleshooting.

8.3.6 Long Term Outlook

8.3.6.1 PROCESS RELIABILITY

Process reliability will be closely related to the level of maintenance. Equipment service lives will be directly related to this factor, as well as site specific factors such as consistency of electrical power, abrasive solids entering the process system, care and frequency of descaling equipment, etc.

8.3.6.2 PLANT INTEGRITY

As a guide, the standard life for chemical equipment is considered to be 11 years (based on US income tax provisions) (Perry and Chilton, 1973). Guideline lives for depreciation in industries of similar nature to an ARD treatment plant appear to be in the range 16 to 18 years. However, these are put forward for accounting purposes, and may not relate well to actual experience. Canadian income tax regulations do not specifically define asset lives, and capital cost allowances for resource industries are in some cases not closely related to expected service lives.

For economic forecasts, it is convenient to think of the plant as having a given service life, following which it will require replacement. However this is not an accurate picture of the actual sequence of events. Usually, various items of equipment will be replaced piecemeal as required, with mechanical/electrical items such as pH probes, pump seals, pump impellers, and agitator impellers requiring more frequent renewal than other items. With reasonable maintenance, the building structure and foundations should last indefinitely. Generally, the process involves at most only mildly abrasive slurries (lime grit) and mildly corrosive solutions (until neutralised), so properly specified and maintained equipment should have an excellent service life.

8.3.6.3 SLUDGE STABILITY

The chemistry of sludge stability was discussed in Section 8.3.3.4. Beyond the chemistry, the stability of the sludge depends on the mode of storage. The sludge should contain some protective alkalinity, which will largely prevent leaching of the precipitated contaminants. However, the alkalinity (calcium carbonate- calcium hydroxide) has a solubility in water, as do the precipitated contaminants. Continual change in the sludge's aqueous environment provides the opportunity for continual leaching of the alkalinity and metals to meet their respective equilibria, and eventually, with the alkalinity gone, pH swings could result in heightened dissolution of contaminants.

This points up the need to dispose of the sludge in a way that immobilizes the water in its immediate vicinity. It should not be too difficult to achieve, since the precipitates are very fine and over lengthy periods should consolidate to an impermeable mass. Local dissolution and reprecipitation of hydroxides may help to "cement" the mass together. Situations to be avoided are those where the sludge may periodically be resuspended and resettled (e.g., shallow ponds), as this provides a change of water and delays consolidation.

8.3.7 Capital and Operating Costs for Typical Installations

8.3.7.1 REVIEW OF AVAILABLE EXPERIENCE

The following information has been taken from published case studies. Costs presented in this section have been updated them using a Canadian dollar Chemical Engineering Index. This index is determined at each date by multiplying the Chemical Engineering Index by the prevailing exchange rate.

Environment Canada

Environment Canada produced a manual on "Mine and Mill Wastewater Treatment" (1975) which included the results of a study by Montreal Engineering Co. The results were presented as a series of graphs from which approximate capital and operating costs could be determined for a given circumstance. In a representative example, a plant size of 164 m^3/h was chosen with an acidity of 2000 mg/L $CaCO_3$.

Brunswick

The Brunswick Mining and Smelting water treatment plant started up in 1978, with a design capacity of 110 m^3/h. The plant incorporated two reactor tanks with a lamella clarifier and sludge recycle; i.e., a high density sludge design. The capital cost at that time was $558,000, and an operating cost of $.56/$m^3$ was experienced, said to be 90% lime and flocculant. No operator was stationed at the plant, and it ran 24 hours per day.

Sullivan

Cominco built a high density sludge plant at Kimberley to treat ARD from the Sullivan Mine and associated tailing impoundments. The need to oxidize significant quantities of ferrous iron with air was a feature of this plant. The plant was started up in 1979. The capital cost for the project was said to approach $10 million for a plant capable of treating 1080 m^3/h. Operating costs were anticipated to be $0.8 million annually for an unstated volume of feed. If we assume 840 m^3/h as "typical", the operating cost would be $0.11/$m^3$.

Les Mines Gallen

The Gallen treatment plant started operations in 1981, with a design capacity of 110 m3/h. The plant incorporated two reactor tanks with a lamella clarifier and sludge recycle; i.e., a high density sludge design. Capital costs were not stated, but operating costs with one operator were $.42/$m^3$, of which lime was 44%, flocculant was 5%, power was 34%, and labour was 16 %. It was planned to run the plant without a full time operator once start up hurdles had been passed.

Equity Silver

Equity Silver have a lime treatment plant that does not use a clarifier with sludge recycle. The plant is capable of treating about 330 m^3/h. No capital cost is presented, as the plant was assembled in part from equipment available on site, but the operating cost has been stated to be $0.70 /$m^3$ for treatment, with an additional $0.15/$m^3$ for pumping (1987 costs). This plant does not have a mechanical thickener, and does not use flocculants.

8.3.7.2 COST SUMMARY

Table 8.3-5 summarizes the capital and operating cost data from the EPS studies, Brunswick, Sullivan, Les Mines Gallen and Equity Silver.

TABLE 8.3-5

SUMMARY OF TREATMENT CAPITAL AND OPERATING COSTS

					Original CAN $		Current CAN $	
Year	Property	Flow m^3/h	Lime Dose mg/L CaO	CE Index	Oper. Cost $/$m^3$	Cap. Cost $1000	Oper. Cost $/$m^3$	Cap. Cost $1000
1975	WPCD-EPS study	164	2000	185.3	0.150	480	0.340	1088
1975	WPCD-EPS study	295		185.3		720		1632
1978	Brunswick	110	7300	259.5	0.559	558	0.905	903
1979	Sullivan	1080	1000	278.5	0.110	10000	0.166	15083
1981	Gallen	110	1850	352.1	0.421	N/A	0.502	N/A
1987	Equity	330	5600	420.7	0.850	N/A	0.859	N/A

Published capital and operating costs for ARD lime treatment plants scaled to 1989 dollars.

8.3.7.3 DEVELOPMENT OF "RULES OF THUMB"

Operating Costs

The operating costs summarized above have been plotted in Figure 8.3-5, with a regression line through the points. There is reasonable correlation of the data with lime dosage, as would be expected since both lime and flocculant dosage correlate with acidity, and lime and flocculant comprise a significant portion of the operating cost. None of the plants require addition of iron to control arsenic or antimony. If this were required, or other polishing treatments were required, operating costs would be higher.

Figure 8.3-5 provides as good a "rule of thumb" as is likely to be obtained. Better estimates would require consideration of site specific circumstances, such as delivered lime cost, staffing requirements, power cost, etc.

Capital Costs

There is little information on which to base capital costs. The costs in section 8.3.7.2 have been plotted against design flow rate in Figure 8.3-6, and it can be seen that the downward curvature that should be expected with economies of scale does not apply. The point for Sullivan seems too high; evidently, it must include costs for research and development, pipelines, and other modifications to surface structures that make it not strictly comparable to the remaining data.

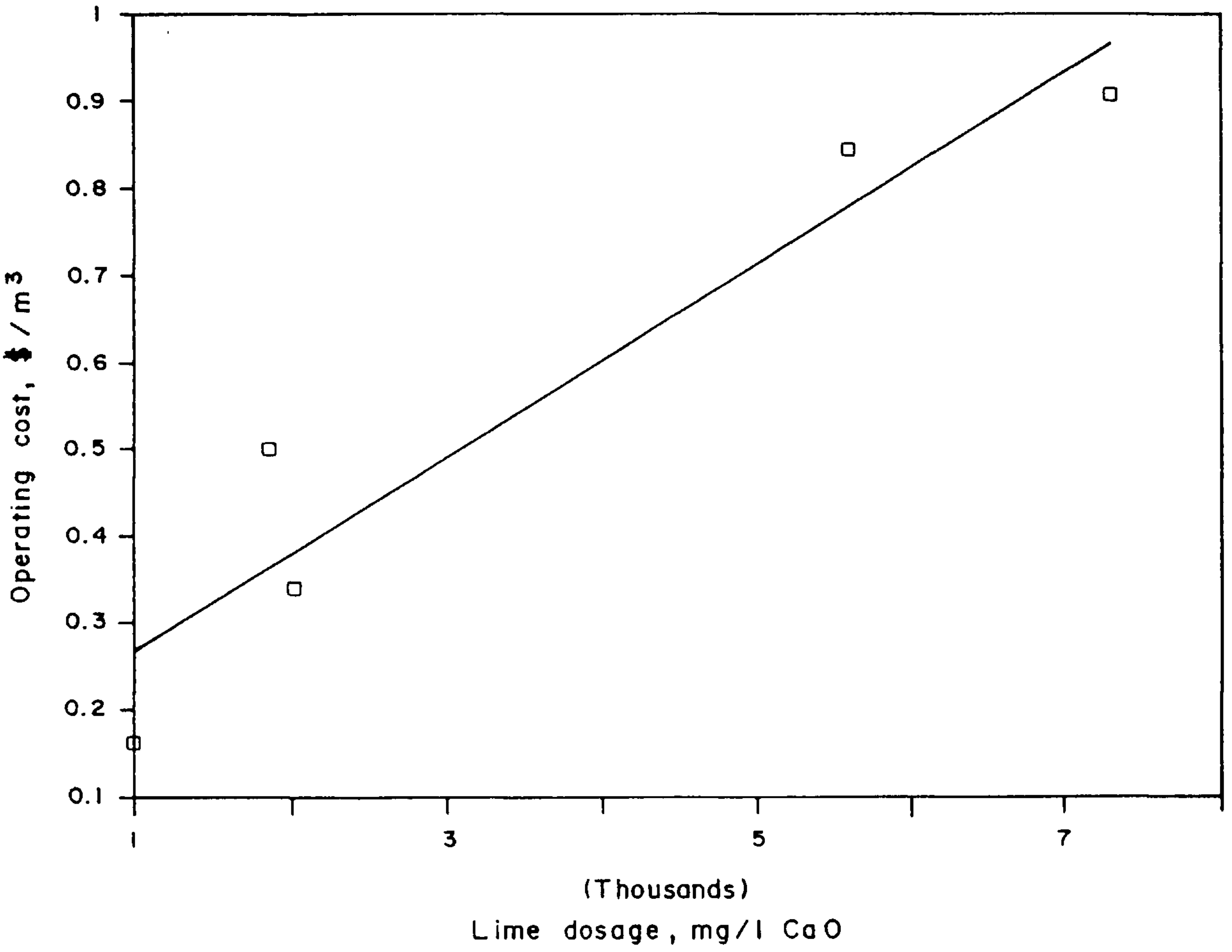


FIGURE 8.3-5
OPERATING COSTS FOR ARD TREATMENT

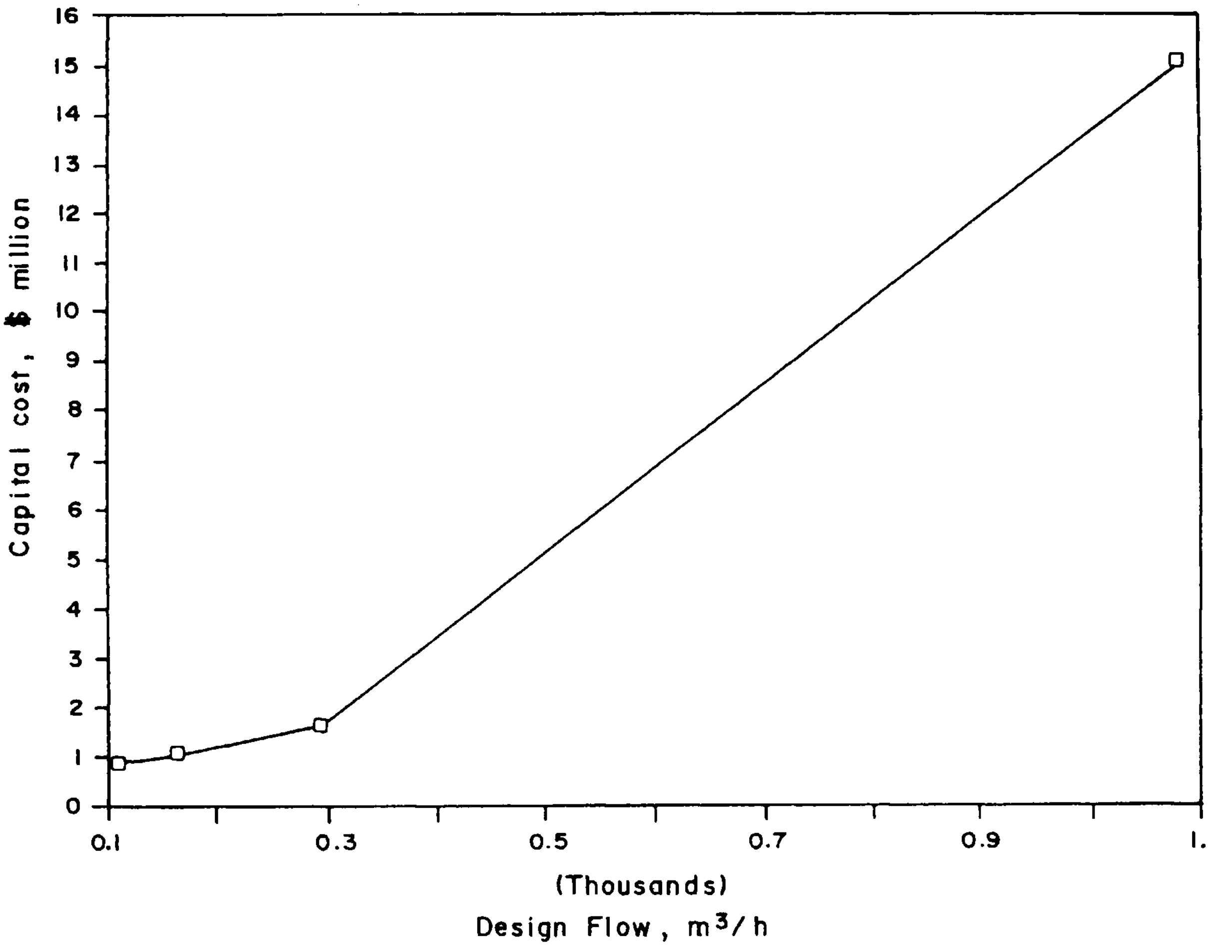


FIGURE 8.3-6
CAPITAL COSTS FOR ARD TREATMENT

For a "rule of thumb", we recommend that the Environment Canada costs or the Brunswick costs be scaled to the appropriate level using the "6/10 rule": the new cost is proportional to the ratio of the new to old flows taken to the 6/10 power.

8.3.8 Future directions for Research & Develepment

Areas where research should be conducted to reduce uncertainties in design and improve process performance and/or economics are:

- Elucidation of sludge chemistry related to stability.
- Improved analytical techniques for low level monitoring, with possible extension to on-line instrumentation.
- Research into causes and cures for calcium scaling on process hardware.
- Improved instrumentation design to reduce susceptibility to scaling, especially pH probes.
- Improved monitoring and control systems
- Develop more effective, cheaper flocculants.
- Develop better design database for selection of agitation and aeration equipment (lessen reliance on vendors).
- Develop better understanding of contaminant precipitation mechanisms (other than straight solubility product determinations), such as coprecipitation, adsorption.

8.4 WETLAND TREATMENT

8.4.1 Overview

Wetlands such as bogs, marshes and swamps are increasingly being recognized for their ability to treat acid rock drainage and other wastewaters. In a mining context, interest in wetlands began when the quality of drainage waters was found to improve after passing through natural bogs and marshes. The vegetation, waterlogged organic soil, and microorganisms, all contribute to the removal from the water of dissolved substances including heavy metals, nutrients and suspended solids. In some wetlands, water pH is also buffered, such that the acidity and dissolved metals content of mine drainage can be significantly reduced after passing through a wetland.

Research to date has shown that the ability of wetland systems to treat wastewaters is dependent on water flow distribution above and below ground, and residence time. These factors are more easily controlled in constructed than in natural wetlands. Also, increasing recognition of the value of natural wetlands as habitat for wildlife, and the need to maintain this habitat in an unaltered state, has stimulated interest in the use of constructed wetlands as an alternative to natural systems. Despite these concerns, natural wetlands continue to be used as incidental polishing areas for acidic seeps at abandoned mines. Over time, constructed wetlands also take on the same habitat functions as natural wetlands.

In British Columbia, the use of wetlands for treatment of acid rock drainage is still in the experimental stages. At Mount Washington Mine on Vancouver Island, a natural wetland receiving acid drainage from the mine site has been dammed to improve water distribution and residence time, and water quality is being monitored by Environment Canada (Ferguson, 1986). Other sites at which the use of wetlands for metals removal and ARD treatment have been considered include a natural wetland at Equity Silver Mine near Houston, B.C. receiving mine drainage and constructed wetlands proposed at Cinola Gold Mine on the Queen Charlotte Islands, (City Resources Ltd., 1988).

Wetland treatment has been included in this guide as a promising technology that is still under development. Detailed how-to information is not provided pending additional R&D werk. Nevertheless, because of the results obtained with wetland systems treating ARD elsewhere (Eger and Lapakko, 1988; Brodie et al., 1988), and the potential advantages of wetland systems particularly for mine closure and reclamation, a description of the treatment processes, advantages and disadvantages, preliminary design criteria, and research needs are presented in the following sections. In general, wetland systems are probably best suited to weakly acidic, low hydraulic flow situations or for the final stage of a staged treatment process as a polishing step for treatment of ARD. (Emerick and Howard, 1988). A number of case studies of wetland systems used to treat mine water were presented at a conference held in Pittsburgh in April 1988 (Stillings et al, 1988; Brodie et al, 1988(a); Stark et al, 1988; Emerick et al, 1988; Hiel and Kerins, 1988).

8.4.2 Wetland Treatment Processes

Water treatment in wetlands is brought about by a variety of interrelated processes, including flocculation and settling of suspended particles, adsorption onto the surfaces of submerged plant parts and organic debris, absorption via the microbial community and plant roots and shoots, and microbially and geochemically mediated chemical transformations such as denitrification and reduction-oxidation. Sulphate reduction and the associated creation of alkalinity, as well as pyrite formation, have been documented in constructed wetlands (Hedin et al, 1988). Possibly the most important mechanisms are associated with the living components, including the bacteria and fungi, as well as the larger plants and the organic matter produced by them. Since the rate of biological transformation is influenced by temperature, treatment effectiveness of wetland systems is seasonal, with maximum treatment taking place during the summer and minimum treatment during the winter.

8.4.3 Advantages of Wetland Treatment Systems

The advantages of wetland treatment systems relative to other treatment technologies are summarized as follows:

- adaptability of wetland systems to acid drainage (pH as low as 2.7) and elevated metals content is high;
- ability of wetland systems to improve the quality of acid mine drainage is well documented in the northeastern and southeastern, and to a lesser extent in the western United States;
- capital costs of natural wetland systems are relatively low;
- operational and maintenance costs of constructed wetland systems are relatively low;
- wetland systems are generally self maintaining and require little or no operator supervision;
- wetland treatment systems also provide additional environmental benefits such as wildlife habitat and flood control.

8.4.4 Disadvantages of Wetland Treatment Systems

The following are disadvantages of wetland treatment systems relative to other treatment technologies:

- in mountainous terrain where sufficient flat or gently sloping land is unavailable, capital costs of constructed wetland systems are elevated due to earth moving requirements;
- land area requirement per unit flow of water to be treated can be large (typically around 50 m^2/m^3d^{-1});
- treatment during winter is reduced;
- impacts on wildlife are still unknown;
- in British Columbia to date operating experience is lacking and treatment performance data are limited to one experimental system using a modified natural wetland (Mount Washington).

8.4.5 Preliminary design considerations

Surface area requirements:

Assessment of whether to use a wetland system at a mine will usually be based on surface area requirements for a given AMD flow regime balanced against the availability of suitable land in the vicinity of the mine site. Information requirements to determine the necessary wetland surface area include:

- estimated mean and maximum AMD flows; and
- estimated mean and maximum concentrations of metals and pH in the drainage water to be treated.

Ideally, AMD flows can be minimized through appropriate water management methods discussed elsewhere in this guide.

Based on wetland designs employed for ARD in the United States, area requirements of 1.4 to 78.5 m^2 for every m^3/d of flow are in use (Brodie et al, 1988; Kleinmann et al, 1986). The wide range reflects both the design flow regime (average or maximum flow) and the water quality (i.e. the degree of treatment required).

Area requirements can also be estimated on the basis of mass loading and pH. Preliminary guidelines for Fe and Mn treatment where desired effluent Fe = 3 mg/L and Mn = 2 mg/ℓ or less, are recommended by Brodie et al. (1988) as follows:

Fe: 2 m^2/mg.min <pH 5.5> 0.75 m^2/mg.min

Mn: 7 m^2/mg.min <pH 5.5> 2 m^2/mg.min

For example, for Fe removal from an average flow of 113 L/min of pH 5.6 AMD containing 50 mg/L Fe and 15 mg/L Mn, the required treatment area would be:

Fe: (0.75 m^2/mg/min)(113 L/min)(50 mg/L) = 4237.5 m^2

Mn: (2 m^2/mg/min)(113 L/min)(15 mg/L) = 3390 m^2

and therefore the required wetland area would be approximately 4200 m^2 or 0.42 ha.

Similar guidelines for other metals such as Cu and Zn, for example, which are a concern in British Columbia, are not yet available.

Since ARD flows tend to vary with season, wetland area requirements should be based on peak expected flows. The above treatment area estimations do not incorporate storm flow conditions, and consequently the size of the wetland would have to be increased and berm design upgraded if necessary to accommodate high flows. This approach is conservative in that a significant safety margin is provided during summer, when flows are lowest.

Other important design considerations:

Other design considerations for constructed wetlands include soil quality and depth, water depth, plant species, and flow distribution. Where natural wetlands are available (and approved by the regulatory agencies to be adapted for polishing ARD), flow distribution is the primary consideration. Available information on each of these factors is summarized briefly below.

Soil quality: The plant-substrate-microbial complex required for AMD treatment can develop in almost any soil type, from clay to pea gravel to natural wetland soil (Brodie et al. 1988a). The main concern is that initial soil fertility must be sufficient to support plant growth. Microbial inocula are unnecessary.

Soil depth: Soil depth of 0.3 m or more is recommended.

Water depth: Average water depth should generally be less than 0.3 m. Depth tolerances of the available emergont aquatic vegetation can be observed at the vegetation donor sites or determined through simple pot tests. Since the actual bottom profile may vary considerably, actual water depth will also vary. Water depths may therefore range from ≤ 0 (subsurface flow) to 1 m or more in some places, and plant species composition may vary.

Vegetation: The most practical approach to species selection for AMD wetlands is to start out with a wide variety of species obtained from natural wetlands near to the mine site, including aquatic mosses and emergent reed like species. Where natural wetlands are not available nearby, transplants would have to be imported from a more distant location. The "rule of thumb" is to transplant while the plants are dormant (late November through late March); the nutrient reserves in the plant roots are greatest during this period.

Flow distribution: Short-circuiting of water flow through a natural or constructed wetland reduces the exposure of the drainage water to the treatment environment. Therefore flow distribution is recommended. Short circuiting can be prevented in constructed wetlands by providing a length to width ratio of at least 10:1. In natural wetlands, existing channels can be re-routed by using earthen or log baffles to create a low velocity sheet flow. Dense vegetation and variable bottom topography, particularly if high points are oriented perpendicular to the direction of flow, also assist in distributing water flow and preventing short circuiting.

A number of papers on wetland treatment process and design were presented at a conference in Pittsburgh in April 1988 (Sencindiver and Bhumbla, 1988; Samuel et al, 1988; Wieder, 1988; Hedin et al, 1988; Brodie et al, 1988b).

8.4.6 Research needs

Wetlands for ARD treatment in British Columbia can be considered a "candidate technology" at present, promising but not developed to the point of routine installation. Further research on wetlands for ARD treatment in British Columbia can probably best be applied to situations such as small, weakly acidic seeps or other limited discharge sites where polishing for removal of slightly elevated metals and sulphate concentrations is desired.

Much of the basic information developed for wetland applications in other parts of North America is transferrable to British Columbia, such as the overall approaches to the assessment of wetland treatment capability and structural design concepts.

The specific challenges facing wetland application in British Columbia are related to the greater diversity of environments in which wetlands might be used. Mines in this province exist or are proposed from sea level to high mountain sites. Climate regimes range from arid to very moist. Such diversity tends to preclude the transferability of results between sites in different climatic and geological zones. Consequently, successful application of wetland systems for ARD treatment in British Columbia will need to be based on information generated in pilot programs in representative climatic and geological locations.

The following specific research topics are recommended:

1. An inventory of existing natural wetland systems that are exposed to natural or mine-unrelated acidic drainage;
2. Pilot scale investigation of hydrologically modified natural wetlands, or constructed wetlands, for treatment of actual or simulated ARD at a moist and dry mine site;
3. Research on natural or experimental sites should attempt to characterize such factors as:
    - hydrology and hydrogeology, including retention time, exfiltration and evapotranspiration, and subsurface flow;
    - plant, periphyton and microbial communities, including surface area available for microbial growth and seasonal trends;
    - water quality and geochemistry, including seasonal trends in acidity and metals cycling; and temperature dependent rate constants; and
    - combination of wetland and other ARD treatment techniques.

8.5 OTHER TREATMENTS

For the treatment of acid rock drainage various techniques have been proposed that are in the conceptual development stages.

Both the use of zeolites and forest waste products have shown the potential for limited use in treating mildly acidic rock drainage.

Zeolites are natural minerals consisting of aluminosilicates with a porous structure that permits the adsorption and desorption of molecules. The mineral structure is negatively charged and the charge is balanced with positive charges from alkali or alkaline earth compounds, such as calcium and sodium, within its pores.

Since zeolites are natural cation exchangers which have the potential for removing metals from waste water, they can be used in treating mine drainage.

Although some of the zeolites are highly selective for certain metals the extent of trace metals removal is dependant on the presence of competing cations, decreasing pH and increasing flow rate (Lapakko and Eger, 1981). Since they are only moderately stable in mildly acidic environments their use should be restricted to the polishing of treated drainage.

When the zeolites dissolve in acidic water the adsorbed metals would be released back into the environment.

The cost of the option is dependant on the supply and source of zeolites used. There is both natural and synthetic zeolites commercially available.

The use of forest waste products has also been studied for treatment of acid rock drainage runoff (Lapakko and Eger, 1981).

Forest waste products are diverse and are readily available. The types of waste products from forestry activities include: twigs, branches, stumps, roots, bark, slabs, chips, sawdust, and veneer cores. All forest products are composed of approximately 50% cellulose. Cellulose undergoes organic oxidative processes to give the wood products a weak acidic ion exchange characteristic. The product is therefore an adsorbent material which is able to remove cations from solution. Wood chips and sawdust have been tested and have demonstrated the ability to remove metals from solutions (Lapakko and Eger, 1981). Sawdust has demonstrated the ability to reduce sulphate and iron concentrations in a stream containing acid rock drainage (Hedin, 1988). The processes involved in the reduction of sulphate and iron in forestry waste products are similar to those in a wetland system (Section 8.4.2).

Forest products are not the most effective means of removing metals from solution but are advantageous in that thay are readily available and inexpensive. Also, once the adsorptive capacity is reached the wood waste could be removed and be disposed of relatively easily and recycling of the metals may be an option.

Since zeolites and forest waste products are not the most effective methods of removing metals (Lapakko and Eger, 1981) their use would be dependant on the availability and cost of other materials, the quality of mine water that is to be treated and the effluent criteria that must be met. Since both techniques show some promise in the treatment of mildly acidic drainage (Lapakko and Eger, 1981) further research would be required to determine the effectiveness of each technique and to demonstrate its use in a full scale operation.

CHAPTER 9: EVALUATION OF CONTROL OPTIONS

9.1 INTRODUCTION

The objective of ARD prevention and abatement is to meet environmental criteria in the long term in a stable condition and at minimum cost and maintenance. The technique or measure most suitable to achieve this objective is determined by a number of site specific conditions.

The suitability and overall long-term performance of any control measure is a function of the longevity of the effectiveness of the control techniques, including its long-term stability, or ability to resist disruptive forces. Certain control or abatement measures have a limited period of effectiveness. These measures require constant operation or maintenance to achieve the control objective. The successful performance of certain control measures may be more susceptible to the disruptive forces of nature than others. The successful long-term operation of any measure is dependent on its ability to continue to abato ARD and also to resist the action of natural disruptive forces.

The evaluation of available control options must consider the long-term performance, risk of failure, environmental impact and cost of each option. In many projects the most suitable option is readily apparent. In more complex situations where more than one option is available, and it is not readily apparent which of these is most suitable, all the relevant factors need to be considered in the evaluation of the options. It is important to be able to identify the risk associated with each control measure and to evaluate the available options in terms of both risk and cost.

The various available options are described in Chapters 6.0 to 8.0 of the Guide. Included in the description of each measure is a discussion of the period of effectiveness of each option. The suitability of each measure for long-term control of ARD and the relative importance of stability considerations for each measure are summarized in this chapter. The natural disruptive forces that threaten long-term stability, the risk of failure and a method for evaluating available options are discussed in this chapter.

9.2 LONG-TERM PERFORMANCE OF CONTROL OPTIONS

Chapters 6.0 te 8.0 describe the available options for control of acid generation, control of acid migration, and collection and treatment of ARD. The control measures described in those chapters are summarized here in Table 9.1-1 with a brief description of the longevity of each option and its vulnerability to long-term disruptive forces.

TABLE 9.1-1

QUALITATIVE DESCRIPTION OF THE LONG-TERM PERFORMANCE OF CONTROL OPTIONS

	Control Measure	Long-Term Performance
1.0	**Control of Acid Generation**	
1.1	Conditioning of tailings/ waste rock	Can be but generally not suitable as a long-term measure on its own. Can be used to improve the long-term stability of the waste.
1.2	Waste segregation and blending	May be suitable for long-term control. Not directly vulnerable to long-term stability.
1.3	Bactericides	Suitable for partial, short-term control. Generally used in conjunction with other forms of control. Not directly vulnerable to long-term stability.
1.4	Base additives	May be suitable for long-term control. Not directly vulnerable to long-term stability.
1.5	Water cover	Suitable for long-term control. Man-made impoundments dependent on long-term stability.
2.0	**Control of Acid Migration**	
2.1	Diversion of surface water	Suitable for short-term control. Maintenance required in long term.
2.2	Groundwater interception	Suitable for short-term control. Maintenance required in long term.
2.3	Covers and seals	Suitable for long-term control. Effectiveness entirely dependent on long-term stability.
2.4	Placement methods	Suitable for short-term control. Maintenance required in long term.
3.0	**Collection and Treatment of ARD**	
3.1	Chemical treatment	Suitable but not ideal for long-term control due to high operation and maintenance costs. Vulnerable to long-term disruptive forces.
3.2	Wetland treatment	May be suitable for long-term control. Vulnerable to long-term disruptive forces.

9.3 LONG TERM STABILITY OF WASTE FACILITIES

Waste facilities can usually be designed to adequately achieve the design and abandonment plan objectives at the time of close-out. However, in the long term, the waste deposit and its control structures are subject to two classes of disruptive forces.

1) Short duration extreme events such as floods, fires, earthquakes and tornadoes which apply forces to the structures in excess of values for which they were originally designed.

2) The slow but perpetual action of forces which bring about deterioration, such as water and wind erosion, frost action, the weathering and chemical change of wastes, covers liners or structural materials, and intrusion by roots, animals and man.

Under the action of these forces, failure is inevitable within geologic time. Periodic maintenance can serve to repair the ravages of many of the perpetual forces. For a well designed facility the required interval between maintenance may be decades or centuries, imposing minimal cost on future generations. Remedial measures may be required after extreme events. The level of periodic maintenance and risk of remedial action which may be appropriate to pass on to future generations must be balanced by the prosent value to society of the resource. It is necessary to document the required maintenance and schedule in detail for the benefit of future generations.

During their evaluation of the long term stability of uranium tailings structures and surfaces, Steffen Robertson and Kirsten (1986a, 1986b), developed a summary of potential causes and associated risk of long term instability of 'generic' Canadian tailings impoundments surfaces. The conclusions from these evaluations are summarized in Robertson & Clifton (1987).

9.3.1 Extreme Events

Because of the long period of interest, the likelihood of extreme events is proportionately large. This likelihood is determined from probability:frequency relationships based on the historical record of events.

A period of record covers an interval during which particular conditions, e.g., climate, applied. Evidence suggests (McInnis, 1985), that climatic cycles occur regularly, varying from relatively minor 30 year cycles to major glacial and inter glacial cycles of tens to hundreds of thousands of years. The weather in several years, or a few centuries from now is unknown (Hare and Thomas, 1974). While it is questionable whether man has altered world climate to date, the potential for material alteration increases and his influence may rival or overrule natural climatic changes in the early twenty-first century or sooner. While the nature of the changes are unknown, it is certain that the extremes to which the impoundments will be subjected will be greater than that predicted from existing records. While it is difficult to quantify the likely charge in extreme events, the designer of long-term structures should consider the consequence of such charges on the structures.

High Precipitation and Floods

Large precipitation events represent one of the most likely causes of waste facility, particularly tailings impoundment failure. Failure during such events are also likely to result in large losses of tailings or waste fines to the environment.

Methods for the estimation of high precipitation events, and for the calculation of the resulting flood flows are well developed for dam design purposes. Current practice is usually to design operating facilities to withstand the one in two hundred year precipitation event. Design for the Probable Maximum Precipitation (PMP) is probably more appropriate for long term conditions.

Earthquakes

Dynamio loads, due to earthquakes, may result in the liquefaction of low density saturated tailings or uncompacted, saturated portions of granular embankments or embankment foundation materials. Failure of the El Cobre tailings dam in Chile, (Dobry and Alvarez, 1967) and Mochi Koshi tailings dam in Japan, (Okusa and Anma, 1980), are ample demonstration. New dams can be designed to appropriate standards. In populated areas, it may be appropriate to design for the long term to cope with the maximum credible earthquake. Methods of earthquake loading probability estimation is well developed for dam design.

Numerous older tailings impoundments have been constructed in a manner which make their embankments susceptible to liquefaction during extreme earthquake events. Abandoned impoundments of this type, if located in an earthquake potential area, and if the consequence of failure is large, may require remedial works to render them stable in the long term.

High Winds and Tornadoes

Tornadoes, tropical cyclones and low pressure systems are sources of high winds. Risk of failure from the last two sources is associated more with the precipitation that accompanies them than from the high winds themselves. Tornadoes, though the most destructive of all winds, are of such a short duration that they do not have a significant potential of effecting stability (Kolousek, 1984).

Forest Fires

Forest fires are expected to occur several times in the period of interest. Of themselves, they do not pose a significant threat to waste facility stability. However, loss of vegetation cover may lead to accelerated erosion by wind and water.

9.3.2 Perpetual Disruptive Forces

Erosion

Erosion may occur as a result of either wind or water action. Both are potentially severe causes of instability of surfaces and covers.

- Wind Erosion

Wind erosion has been observed to be a major release mechanism at some existing waste impoundments. Control of this mechanism, for the long term, depends on the successful establishment and maintenance of a wind erosion resistant cover, such as vegetation, waste rock or surface crusting. Methods for the determination of the wind erosion potential and release rates of a particular cover type are reviewed by Steffen Robertson and Kirsten (1986a). They conclude:

1) Unacceptably high wind erosien rates occur on tailings surfaces and soil covers (without gravel) unless a well developed vegetation cover is established. Loss of vegetative cover (due to forest fires, flooding, or salt migration) could result in unacceptably high rates of wind erosion releases.

2) Till, with a substantial gravel percentage, and rock waste will form effective wind erosion resistant layers when placed in thin (600 mm) cover layers.

- Water Erosion

This is probably the single most severe cause of impoundment instability. Erosion can take the form of flood erosion of the diversion works, or sheet and gully erosion of the impoundment surface and embankment slopes.

i) flood erosion

A substantial portion of total erosion occurs during extreme precipitation and flood events. The probability of failure will depend on the criteria used to design the structure, and the degree of scour, sedimentation and/or blockage which has occurred. Sedimentation, ice, vegetation growth and debris blockage are extremely difficult to avoid in the long term; though easily and inexpensively cleared through maintenance.

Methods for the evaluation of erosion risk, and appropriate methods of design are reviewed in Steffen Robertson and Kirsten (1986a, 1986b). Appropriate control structure design would involve:

1) Design and construction of diversion structures to accommodate the Probable Maximum Flood (PMF) with ample width and size to allow for partial blockage or sedimentation.

2) Use of heavy riprap (or waste rock) armouring along flow channels and on adjacent slopes. Methods for erosion protection design are provided in Walters and Skaggs (1984).

ii) sheet and gully erosion

The most suitable methods for the prediction of sheet and rill erosion are the Unified Soil Loss Equation (Wischmeier and Smith, 1978), and Modified Universal Soil Loss Equation (Williams, 1975). After using these methods for erosion rate estimation, Steffen Robertson and Kirsten (1986a) concluded that:

1) Tailings and bare soil would result in excessive sheet and rill erosion unless the soil contained a high percentage of coarse gravel.

2) Good grass cover does much to control this type of erosion which becomes insignificant with continuous forest cover.

3) Rock waste and cobble riprap are effective controls.

4) Discontinuous cover or periodic cover loss will result in unacceptable erosion rates.

iii) gully erosion

Gully erosion has been observed to be a major cause of instability of tailings surfaces, embankments and covers. The only available method of gully erosion estimation is that proposed by Falk et al (1985). This method is based on limited data for American climatic conditions, does not account for vegetation cover and is considered invalid for long period estimation. Nevertheless, it has been used to demonstrate that the only effective gully erosion control for gravel free embankments and tailings slopes is riprap. This conclusion is in agreement with field experience.

<u>Biotic Activity</u>

i) root penetration

In general root action is considered to have an overall beneficial effect on covers, embankments and other structures in providing a binding effect and resistance to soil erosion, as well as minimizing infiltration. Two possible destabilizing effects have been observed:

1) Roots may penetrate low permeability layers and, on decomposing, provide seepage channels which increase infiltration through covers, or piping in embankments.

2) Extensive root development in moist, permeable drains or drainage layers may ultimately result in clogging of the drains, in a manner similar to that experienced with residential drains.

Methods to reduce the potential for drain blockage include ensuring that the drains operate in a flooded condition and use of large drains with surplus drainage void space.

ii) burrowing intrusion

Burrowing intrusion by insects and animals have the potential, in the long term, of significantly altering the permeability of low permeability capping layers.

Intrusion by man, for example, to obtain tailings for use in construction, has proved to be a material transport mechanism at some acid generating waste sites. Vehicle and large animal traffic can also be a major cause of erosion. The prevention of intrusion by man, in the long term, can only be achieved through institutional control and enforcement.

Frost action

Recent studies and experience regarding frost action in tailings impoundments (Knight and Piesold, 1986; Geocon, 1986; Steffen Robertson and Kirsten, 1987), have demonstrated that the effects of frost on the engineering properties of tailings and their containment structures can be large. In those areas where continuous or discontinuous permafrost develops, and in areas of severe winter cold, frost action may be a major cause of long term instability.

The effects of freezing temperatures on tailings impoundment stability can be divided into two broad groups.

i) annual ice accumulation

Water flow in channels or drains may freeze in successive layers resulting in large accumulations of ice in a single winter, ice accumulations may result in blockage of the diversion structures or outlet works, with a consequential risk of erosion along the displaced flow channel during the early spring melt. Freezing of drains may result in a build up of pore pressures in embankments resulting in slope failure.

Ice accumulation can and does occur during tailings placement, where tailings are discharged onto beaches. Depending on the rate of rise and the sequencing of the beach development, annual accumulations may be many metres thick. Incumbent layers of tailings serve as a thermal insulation layer resulting in successive annual accumulations of frozen tailings. These layers of frozen tailings prevent drainage and hence both the dissipation of pore pressures and the consolidation of the tailings.

As a result, large consolidation settlements may occur after mine closure. Such settlement will affect the drainage pattern on the surface of the impoundment and may result in cracking of any cover layers placed on the tailings.

ii) seasonal frost penetration

Knight and Piesold (1986) found that the effect of freezing low density tailings is to create a lattice work of ice crystal with consolidated tailings in between. The effect on the engineering properties of the tailings, (Robertson, 1987), is to increase the permeability of the tailings at the same time as increasing the density. Seepage and consolidation rates (after thawing) are therefore increased. Frost susceptible cover materials subjected to freezing will develop ice lenses and a fissured structure which, on thawing, increases tho permeability of the cover. Where the cover is designed to limit infiltration this represents a severe failure mechanism. Frost penetration may also block subsurface drains, preventing drainage and causing pore pressure increases.

With annual accumulations of ice, there is an associate desegregational frost heave. Conditions have been identified where up to 200 mm of heave can occur annually (Steffen Robertson and Kirsten, 1987). This seasonal heave is uneven, depending on surface and deposit conditions. Heave, with its consequential effects on drainage, wind and water erosion and cover disturbance, further effects differential frost heave conditions. The end result is a hummocky, irregular surface with greatly disrupted drainage and cover layers. Small variations in seasonal temperature, snow cover and vegetation cover conditions will materially effect rates of aggradation and degradation of the frost. Frost induced creep (selifluction) also occurs on steeper surfaces such as embankment and waste pile slopes.

Weathering

The chemical and physical change of cover materials due to weathering may result in long term degradation of the protective layers. Factors to consider include the effects of alternate wetting and drying, alternate temperature changes, ultraviolet radiation, and chemical reaction with pore fluids. Methods for the selection and evaluation of riprap durability are reviewed in Steffen Robertson and Kirsten (1986a, 1986b). Long term durability of natural materials for use as liners has been studied by Clifton Associates Ltd. (1986), Uziemblo et al (1981), Nasiatka et al (1981) and Buelt and Barnes (1981), these results should be applicable to the durability of soil covers as well. Golder Associates (1984) and Buelt and Barnes (1981) have evaluated the durability of synthetic materials. An important consequence of weathering for certain acid generating waste rock types may be that disintegration of rock causes fresh sulphide minerals to be exposed and hence continued acid generation potential.

9.4 RISK OF FAILURE

9.4.1 Introduction

Probabilistic analysis is becoming an increasingly popular tool for assessing the environmental effects and risks associated with waste management. Risk assessment can be used in evaluating complex mine projects proposed in environmentally sensitive areas where the consequence of failure is severe. This technique would not be used extensively for proposed mines and very rarely for existing operations. Various levels of risk assesement may be applied, ranging from simple qualitative studies to identify areas that require special attention, to detailed quantitative studies may allocate a probability of occurrence to specific events.

A probabilistic analysis employs the same mathematical models as the deterministic analysis. However, the approach explicitly accounts for uncertainty in the input parameters by considering a probability distribution, rather than a single value. The mode is evaluated many times with new values selected for the input parameters. The parameters used in each trial are randomly selected from the appropriate distributions using a computerized random sampling technique. With this approach, many individual estimates of the "effect" are obtained by simply performing the number of trials desired. The results of such a probabilistic analysla are conveniently summarized by ordering the values into a frequency plot and calculating desired summary statistics (e.g., mean, standard deviation, percentiles, etc.). It has the advantage over the deterministic approach in that the full range of uncertainty in the parameter values may bs incorporated into the analysis in an attempt to avoid the usual one-sided bias in the input specifications and results.

It is necessary to consider not only probability of failure but the consequences of failure. The concept of 'risk' and 'hazard' need to be considered. Risk may be defined as a compound measure of the probability and magnitude of adverse effects, or

Risk = Uncertainty x Damage

Other definitions of risk are 'the chance of encountering harm or loss' or the 'degree of probability of such loss' (Federal Emergency Management Agency 1984). The dictionary defines hazard as a 'source of danger'. Hazard, therefore, simply exists as a source. Risk includes the likelihood of conversion of that source into actual delivery of loss, injury or some form of danger. Also,

Risk = Hazard/Safeguards

This implies that risk may be kept as small as desired by increasing the safeguards. As a matter of practical reality, however, risk can never be brought to zero (Federal Emergency Management Agency, 1984).

It is necessary to assess the risk of failure of ARD control options in order to identify those options associated with high risk, to evaluate the cost vs benefit of different options, and to suitably design and maintain the selected control measure. For obvious reason it is important to identify the high risk options. Less obvious perhaps but equally important is the interaction between maintenance and risk. A lack or neglect of maintenance increases the risk of failure. For example, it is inevitable that soil covers will erode over time. The effectiveness of covers is dependent on their integrity and therefore some degree of maintenance is required which, if not fulfilled, will increase the probability and hence increase the risk. Similarly, it is inevitable that steel will corrode and regular maintenance of steel structures (e.g., in chemical treatment plants) is required to prevent an increase in probability of failure and hence an increase in risk.

The environmental risk and consequences of mine waste management, may be evaluated by means of risk assessment studies. The principles of risk assessment are summarized by Kaplan and Garrick (1981):

> "In analyzing risk we are attempting to envision how the future will turn out if we undertake a certain course of action (or inaction)."

Fundamentally, a risk analysis consists of an answer to the following three questions:

- What can happen? (i.e., what can go wrong?)
- How likely is that that will happen?
- If it does happen, what are the consequences?

Risk assessment has been developed around four general steps (Van Zyl, 1987):

- Hazard identification and assessment
- Exposure assessment
- Consequence or dose-response assessment, and
- Risk characterization.

Hazard Identification

In the first step, the hazards associated with specific project systems or activities are identified without describing the consequences or risks involved in their potential release. The possible contaminants and/or physical hazards of concern are identified as to the threat they may pose to the environment.

Exposure Assessment

The exposure assessment follows the release of the contaminant from its origin to the receptor. The assessment considers the potential (or probability) of a release, magnitude of a release, the duration of a release and ultimately the concentration of a release received by a receptor. It is necessary to evaluate the transport pathways and concentrations of contaminants reaching the exposed ecosystems.

Consequence Assessment

A consequence assessment evaluates the impact of a release on the ecosystems or organisms of concern, or it addresses the question: if it does happen, what are the conseques? Impacts on terrestrial ecosystems, water quality and aquatic ecosystems in the site area are evaluated in this step of the risk assessment.

Risk Characterization

The last step in a risk assessment is to characterize the risks based on results of the previous steps. This step may comprise either a qualitative or quantitative risk characterization.

9.4.2 Qualitative and Quantitative Risk Assessments

A qualitative risk assessment uses descriptions to identify (or rank) the relative magnitudes in each step of the assessment. An example of a five-point descriptor score is: negligible, very low (or very small), low (or small), medium, and high (or large). These descriptors are defined on a site specific basis for each step of the risk assessment. These descriptors are combined to obtain a characterization of the risk associated with each potential failure mode.

The risk imposed by a specific release, or a combination of independent failure modes in a complex system, can be evaluated qualitatively using probabilistic procedures. A quantity risk assessment is the combination of the probability of a release and the consequence of an exposure. Fault trees and event trees present powerful tools to perform these probabilistic analyses.

A fault tree analysis is a quantitative evaluation of the quantities of the various faults or failure events leading eventually to the calculation of the probability of the main fault event (top event). The potential consequence may be systematically identified with the event tree diagram in which all the possible paths following a top event or initiating event may be traced trough the subsequent event, each path leading to a different consequence (Ang and Tang, 1984).

A fault tree is the graphical development of a top event (major fault) into the union and/or intersection of subevents (faults), 'OR' gates indicate the union of subevents, while 'AND' gates signify the intersection of subevents. When the probabilities are small 'OR' gates can be considered as the sum of two probabilities. For 'AND' gates, the product of the probabilities is used. An example of a fault tree is shown in Figure 9.4-1.

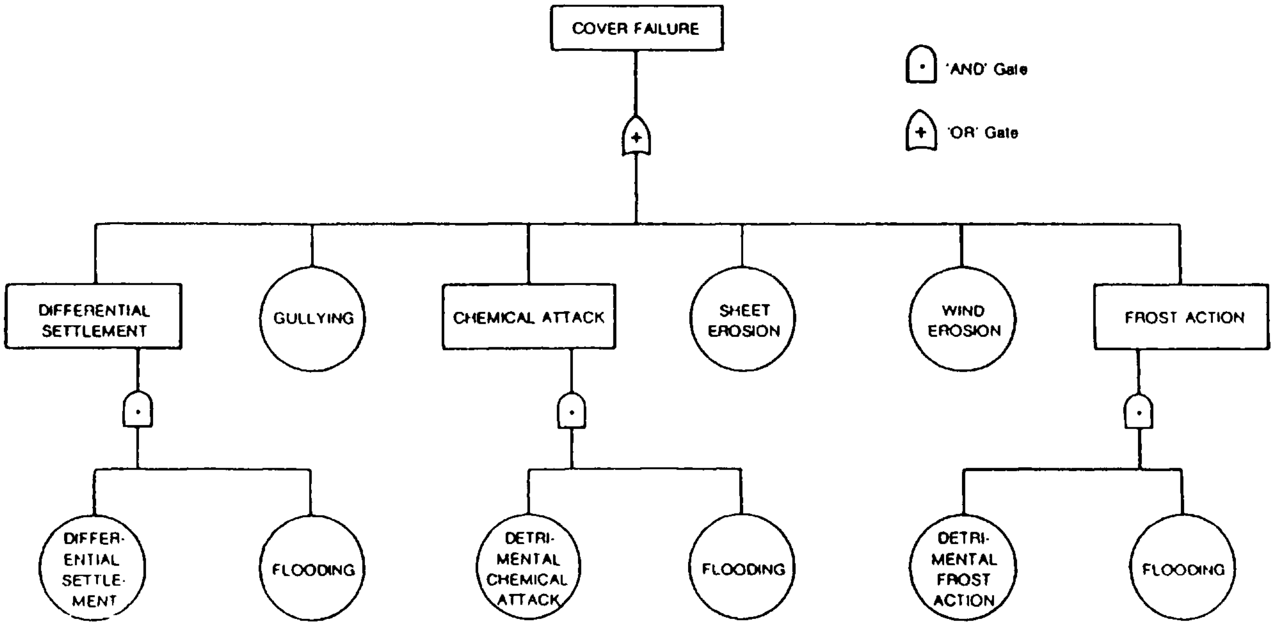


FIGURE 9.1-1
EXAMPLE OF A FAULT TREE FOR COVER FAILURE

In an event tree, the particular consequence depends on the subsequent events following the initiating event. Given an initiating event there may be several 'first subsequent events' that will follow, all of which must be mutually exclusive and exhaustive. Each path in the event tree represents a specific sequence of (subsequent) events, resulting in a particular consequence. The probability associated with the occurrence of a specific path is simply the product of the probabilities of all the events on that path (if the events are independent, an assumption is usually made).

The probabilities used as input to fault trees and event trees can be calculated as discussed above for single failure modes, or by some other objective probabilistic approach. However, in many cases it is necessary to use subjective probabilities obtained through engineering judgement. The use of subjective probabilities does not reduce the rigour of the reliability analysis. It could be used as a basis for sensitivity analysis, where the subjective probabilities are varied over the most probable range andtheir influence on the result investigated.

The advantage of a quantitative risk assessment is that use can be made of the probabilities to compare alternative designs and develop risk management strategies.

The uncertainties and variability in the information and data used can be more clearly evaluated and incorporated in a quantitative risk assessment.

9.5 RECOMMENDED METHODOLOGY FOR EVALUATION OF OPTIONS

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 ARD from a waste dump (Steffen Robertson and Kirsten, 1987) an assessment method proposed by Robertson (1987) was used which considers 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 ARD

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 9.5-1. 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 9.5-2. 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 obtained. The cost/benefit values determined for this project are shown in Table 9.5-2. 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 ARD 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 ARD abatement plan.

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

It should be the objective of all ARD abatement plans to reduce monitoring and maintenance to a minimum commensurate with the available technology and long-term cost. The current state of the art of long-term ARD 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 measure with time (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.

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 options of maintenance in the design of ARD abatement measures. Under these circumstances some of the alternative abatement measures, which might otherwise be rejected has been of too short a duration, become more advantageous. An example is the use of synthetic liners to cover wastes. A thick (2 mm) HDPE membrane liner, appropriately installed as part of composite cover, may be expected to last for 50 to 100 years or more.

TABLE 9.5-1
ABATEMENT OPTION: SYNTHETIC LINER OPTION - ACID ABATEMENT ALTERNATIVES RANKING SYSTEM
(After Robertson, 1987)

Consideration	Description	SCALE					Weighting Factor	POINT TOTAL
		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	x4	20
		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	95-100		
		less secure				more secure		
7. Effectiveness	Reduction of Contamination	0	2	3	4	5	x10	40
TOTAL RANKING POINTS								124

Cost/Benefit = $\frac{\text{Preliminary Total Estimate}}{\text{Total Ranking Points}} = \frac{\$820{,}000}{124} =$ ~~$6,612/Point~~

TABLE 9.5-2

OPTION RANKING ACCORDING TO TOTAL OF RANKING POINTS

(After Robertson, 1987)

Rank	Points	Options	Cost/Benefit $/Point
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

CHAPTER 10: RECOMMENDED APPROACH TO ARD CONTROL

10.1 INTRODUCTION

Certain site specific information is necessary before evaluation of ARD control options can be carried out. To begin with, it is necessary to assess the acid generation potential, based on geochemical test results. The size and status of the project or facility i.e., whether proposed, operating, or shut-down, etc., under consideration is important in determining which control options are practical and feasible. If acid generation is already occurring at the facility, it is important to determine the extent of ARD and what stage it has reached in its evolution. This will have an impact on the effectiveness and suitability of certain control measures. It is necessary to identify whether both short-term and long-term controls are needed, e.g., control during mine operation (short term) and following closure and rehabilitation (long term). The level of ARD control that is required is determined by the impact that ARD would have on the environment and is a function of the quantity and quality of ARD, and the sensitivity of the receiving environment. Once these conditions have been identified for a particular project, the process of evaluation, design, and implementation of ARD control may begin. ARD may be controlled by either preventing the acid generating reactions occurring, preventing the products of the acid generation processes migrating to the environment, or collecting and treating the drainage.

The most desirable form of ARD control is to address the problem at the source, i.e., to prevent or inhibit acid generation from occurring. The approach to ARD control for all projects should therefore begin by investigating methods to control the acid generation process. Measures for preventing contaminants migrating from the waste to the environment and finally collecting and treating ARD should be evaluated in turn if acid generation cannot be adequately controlled. The most effective control may be achieved by incorporating a combination of measures from each of the three categories of control.

Methods of controlling the acid generation process are described in Chapter 6.0 and control of ARD migration, and collection and treatment of ARD are discussed in Chapters 7.0 and 8.0, respectively. A recommended approach to ARD control is presented as a flow-chart in Figure 10.1-1. The recommendations contained in each of Sections 10.2, 10.3 and 10.4 should be read in conjunction with the flow-chart.

10.2 A RECOMMENDED APPROACH TO ACID GENERATION CONTROL

The development of a plan to control ARD should begin by investigating means of controlling acid generation. A recommended procedure for controlling acid generation, as shown in Figure 10.1-1, is set out below:

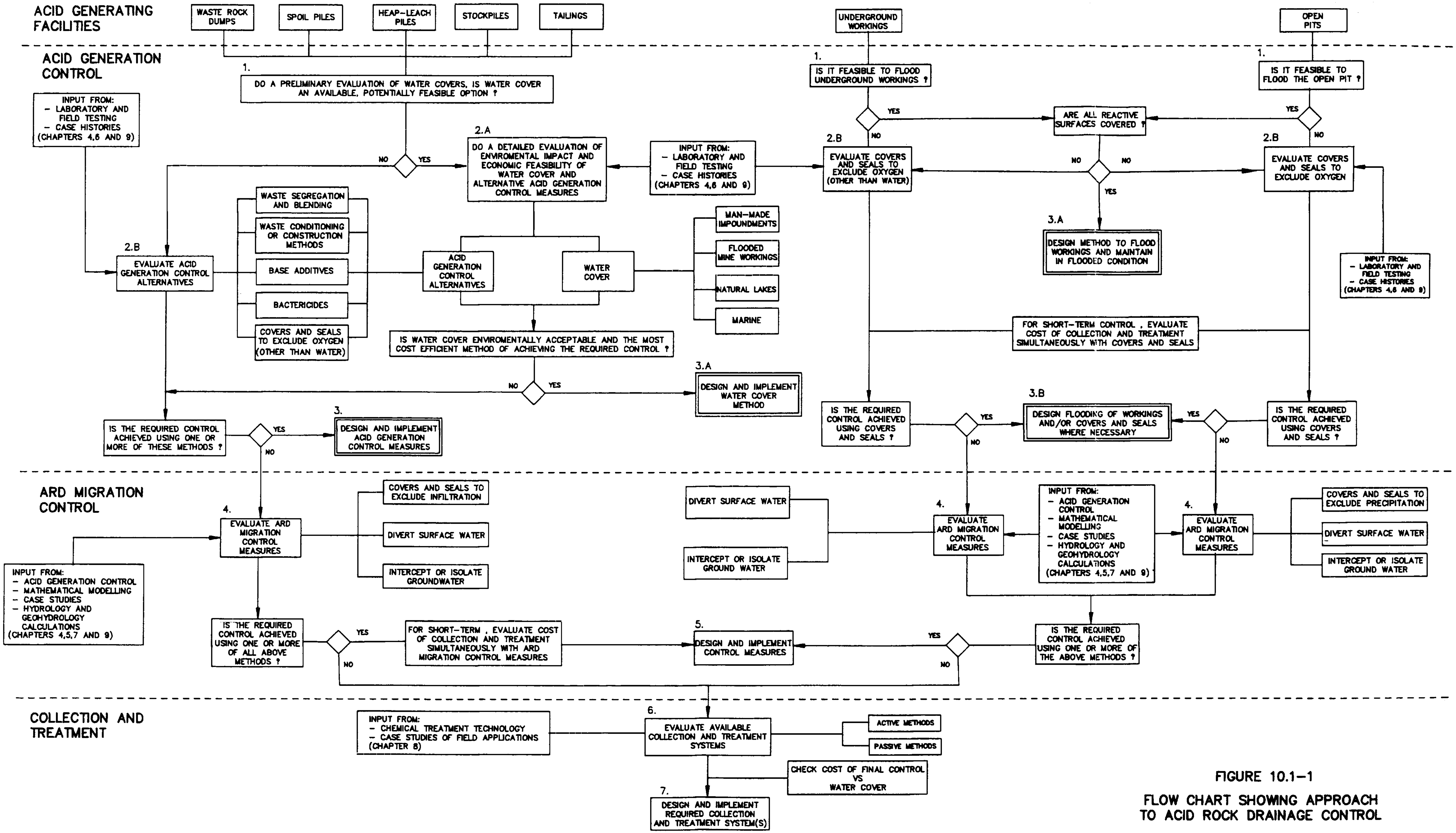


FIGURE 10.1-1

FLOW CHART SHOWING APPROACH TO ACID ROCK DRAINAGE CONTROL

Stage 1

Do a preliminary assessment of the availability of sites for water cover and the feasibility of this method. The purpose of the preliminary evaluation is to determine whether water cover is at all possible, given that this is currently the most promising control measure. Water cover can be provided by means of any of four general methods as follows:

1) Man-made impoundments that maintain the waste under saturated conditions. This includes man-made lakes and saturated soil/water covers (e.g. constructed swamp/ wetland conditions).

2) Flooded mine workings including underground and open pit mines.

3) Waste disposal into natural lakes.

4) Waste disposal into marine waters

The different types of water cover are discussed in detail in Chapter 6.0. Current legislation (e.g. Federal Metal Mining Liquid Effluent Regulations) and the politically controversial nature of lake and marine disposal of mine tailings (acid generating or otherwise) suggests that gaining approval for an application for this method of disposal for tailings will be difficult to achieve (Knapp, 1989). For this reason it is probably beneficial to fully investigate "on-land" means of achieving water cover first i.e. items 1) and 2) above. Lake and marine disposal of tailings should generally only be considered when all on-land options have been exhausted. This may not necessarily be the case for other waste types, e.g. waste rock.

The preliminary evaluation should take into account the existing environmental conditions at the site (base-line), the level of control required, availability of suitable sites, and approximate cost estimates. If the initial evaluation indicates that water cover may be feasible, proceed to Stage 2A, if not proceed to Stage 2B.

Stage 2A

Do a detailed environmental appraisal and economic feasibility study of water cover options and the available acid generation control alternatives. The evaluetion should first determine whether the required control can in fact be achieved using only acid generation control measures (other than water cover). In some cases, it may be necessary to include one or more measures from the categories of acid generation control, ARD migration control, and/or collection and treatment of ARD to achieve the required control. The cost of water cover should be compared to the cost of the alternatives, or combinations of alternatives, that provide the required level of control. Input at this stage will be derived from the results of laboratory and field prediction tests (see Chapter 4.0), some level of baseline environmental measurements (see Chapter 11.0) or conclusions drawn from field observations, and records of material and construction costs. Evaluations should consider long-term performance and stability of alternatives (see Chapter 9.0). If this study shows that water cover is environmentally acceptable and the most cost efficient method of achieving the required control, proceed to Stage 3A. If a combination of alternative measures achieves the required control at lower cost, proceed to Stage 3B. If water cover is not feasible, and alternative acid generation control measures do not achieve the required control, proceed to Stage 4.

Stage 2B

If the initial, preliminary evaluation of water cover indicates that this method is out of the question, the acid generation control alternatives should be evaluated in detail to determine the most efficient combination of measures to achieve the required control. If the required control can be achieved using a combination of these measures, proceed to Stage 3B, if not, proceed to Stage 4.

Stage 3A

If water cover is environmentally acceptable and provides the most cost efficient method of achieving the required control, the design and implementation of this method should proceed.

Stage 3B

If one or a combination of acid generation control measures achieve the required control at a lower cost than water cover, design and implement the method(s).

10.3 A RECOMMENDED APPROACH TO CONTROL OF ARD MIGRATION

When water cover is not acceptable and alternative acid generation control measures on their own do not provide adequate control, measures to control the migration of ARD should be investigated. A recommended approach, as shown in Figure 10.1-1, is described below:

Stage 4

Evaluate available measures to prevent the flow of water into or onto the waste. These include means of excluding infiltration of precipitation, diversion of surface runoff, and interception of ground water. This evaluation may be assisted by input from:

i) The results of Steps 1 to 3B above (Section 10.2), i.e., what further control is required to meet environmental acceptance criteria.

ii) The results of mathematical modelling of, for example, infiltration through soil covers (see Chapter 7).

iii) Data from case histories, e.g., Rum Jungle, Mt. Washington, etc. (see Chapter 7).

iv) The results of hydrology and geohydrology calculations to determine design flows, etc.

v) Geotechnical studies (e.g., availability and cost of construction materials for covers).

vi) The results of geochemical testing to evaluate the effectiveness of specific control measures (see Chapter 4).

If this evaluation indicates that these measures, together with acid generation control, are adequate, proceed to Stage 5. In the cases where only short-term control is being evaluated, and the cases of control for existing facilities, collection and treatment should be evaluated simultaneously with migration control measures. The reason for this is that collection and treatment may be a feasible short-term measure or, for existing facilities, may represent the only practical option. If the results of the evaluation indicate that further control is required, proceed to Stage 6.

Stage 5

Proceed with the design and implementation of the control measures. A sound design will include a contingency or back-up measure in the event that the predicted level of control is not achieved on site. The design and decision making procedure should be integrated with the monitoring programme (see Chapter 11.0).

10.4 A RECOMMENDED APPROACH TO COLLECTING AND TREATING ARD

When collection and treatment of ARD is required as indicated by Stages 1 through 5 above (Sections 10.2 and 10.3), shown on Figure 10.4-1, a plan to collect and treat the ARD should be developed.

Stage 6

Evaluate available treatment systems including both active and passive chemical treatment methods (e.g., chemical treatment plant, alkaline trenches), and natural treatment methods (e.g., wetlands).

Stage 7

Design and implement collection and treatment. A recommended approach to the development of a chemical treatment plan is shown separately as a flow-chart in Figure 10.4-1 and described below:

7.1 Identify maximum and average flows. For proposed projects, assign a probability of contamination to each potential source, and list them all. Then, evaluate the potential for equalization ponds before and after treatment, as these are desirable for minimizing plant size and permitting steady, reliable operation.

7.2 Characterize feed (water quality) as well as possible. For proposed operations, make projections based on mineralogy, as well as testwork effluent quality (see Chapter 4.0). For an existing problem, conduct a sampling program over at least one year (see Chapter 11.0). Estimate the future seasonal quality, and try to determine the requirement for treating new sources that are not currently a problem. Estimate the probable sludge quality and quantity based on the expected ARD chemistry.

7.3 Evaluate any potential for recovery of valuable constituents. Select the simplest process route. Evaluate the sludge disposal options. Underwater disposal in a quiet pond or lake is preferred. The high density sludge process should only be required if sludge disposal is a problem, and there is insufficient pond capacity for clarification. Pressure filtration of sludge should be considered if the sludge has to be trucked off the site. Aeration may be necessary in the treatment process if significant ferrous iron is anticipated. Addition of ferric iron for arsenic control may be necessary if there is insufficient in the feed. Consider special processing options if it is necessary to remove problem contaminants; e.g., Hg, CN^{-x}, etc. Investigate cheap local sources of alkalinity versus purchased reagents, e.g., tailings, fly ash, limestone, etc.; do a preliminary economic comparison. Generally, try to limit the use of mechanical equipment in favour of long term reliability and reduced operating cost.

7.4 Test the proposed concept in the laboratory. Demonstrate the technical feasibility of meeting effluent objectives, determine the required quantities of reagents, demonstrate sludge composition and stability, and develop information for sizing equipment.

7.5 Develop conceptual design. Design the flowsheet, and develop an equipment list. As part of this activity, study the economics of process alternates. Determine the reagent supply arrangements, and design storage and makeup facilities. Decide on the lime slaking method, and determine the need for aeration, clarifier, sludge recycle, and filtration. Determine the type of clarifier. Remember that staged neutralization is preferred over a single vessel. Lay out the mechanical general arrangement. Estimate the capital cost, for budgeting and cost control.

7.6 Complete the detailed engineering, build the plant. This step should be delayed as late as possible so design is based on best available information. Emphasize a simple, reliable, minimum maintenance design.

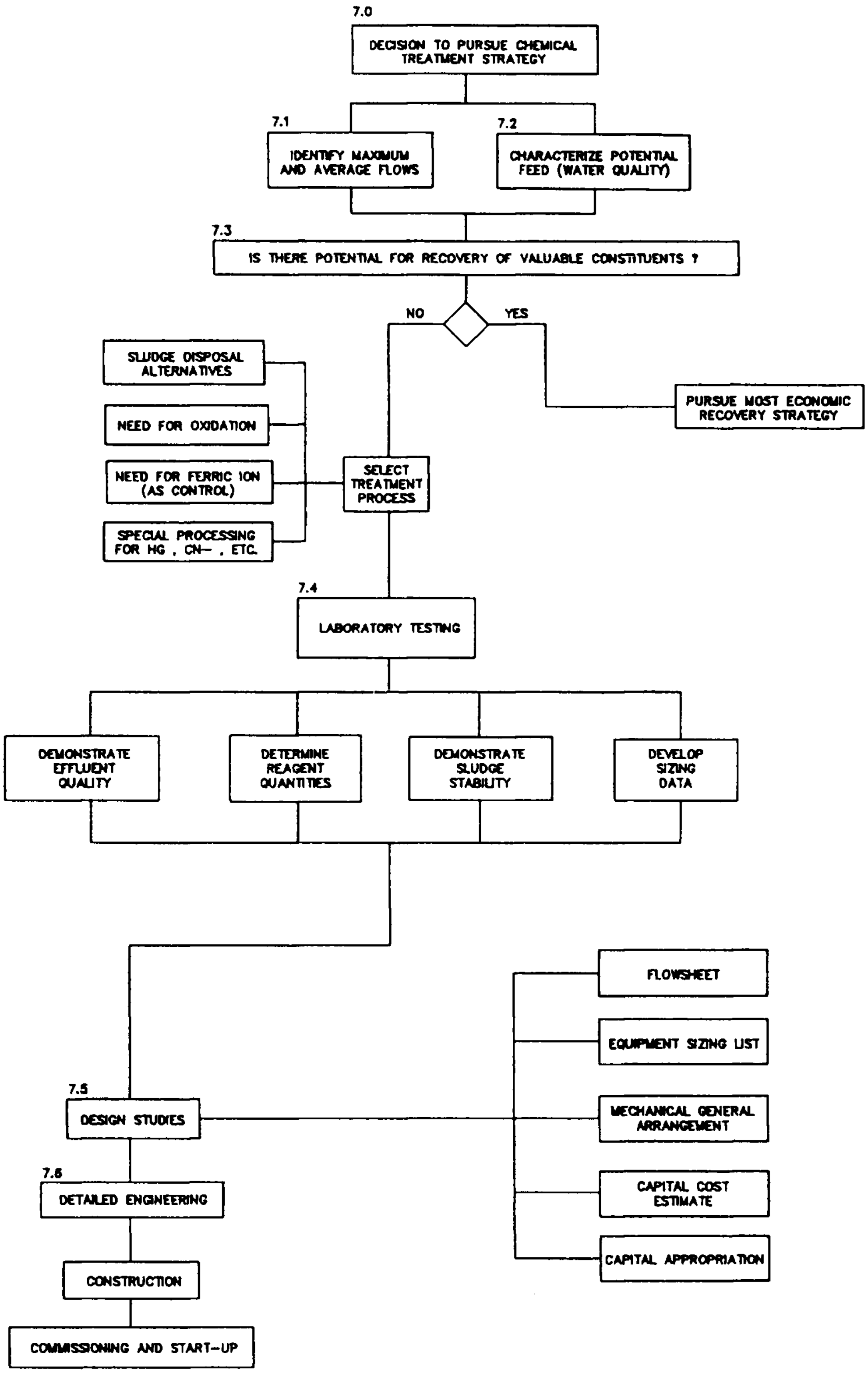


FIGURE 10.4-1
FLOW-CHART SHOWING RECOMMENDED
CHEMICAL TREATMENT PLAN DEVELOPMENT

CHAPTER 11: MONITORING

11.1 GENERAL GUIDELINES FOR MONITORING

In general terms, environmental monitoring in and around a minesite is intended to define baseline conditions and to identify changes in conditions during and after mining. This information is generally used for decision making regarding mitigation and reclamation strategies.

The environmental conditions monitored typically include physical processes such as water flow and geotechnical stability, chemical characteristics such as water quality, along with biological response and impacts such as productivity. The major objective of a monitoring program in the acid rock drainage context is to monitor the effectiveness of the prevention/control/treatment techniques and to detect at the earliest point in time if the techniques are unsuccessful. A flowchart outlining a proposed environmental monitoring approach is presented in Figure 11.1-1, addressing Pre-operational, Operational, and Closure Phases of a mining project.

In the pre-operational phase (Figure 11.1-1), baseline monitoring defines existing environmental conditions of the physical, chemical and biological aspects of the area. This information leads to the identification of areas that are particularly sensitive to changes in environmental conditions and provides essential data to allow the assessment of changes or impacts caused by each component of the mine and mining activities.

In terms of acid generation, testwork is conducted in the pre-operational phase to determine the potential of each waste material from the various mine component to generate net acidity and acid drainage. Each of the potentially acid generating materials may be further tested to determine the rate and duration of acid generation and its associated water quality. The design and testing of the required control and treatment techniques may also be conducted in this phase. The mine plan is adjusted in order to reliably implement the control/treatment techniques, to reliably eliminate acid drainage, and to minimize combined costs of environmental protection, mine construction, and operation.

Using the baseline information and the final mine plan, a monitoring program is established for mine operation. A detailed flowchart of a proposed integrated on-site/off-site monitoring program for operation and closure is presented in Figure 11.1-2. Two types of monitoring stations have been defined for this approach: an effluent discharge point and the receiving environment. An effluent discharge point is generally but not necessarily located on the mine property (on-site) while the receiving environment stations will generally but not necessarily be off-site.

In the proposed program, monitoring stations are established in or near all environmentally sensitive areas potentially affected by the developmont including both surface water and groundwater stations.

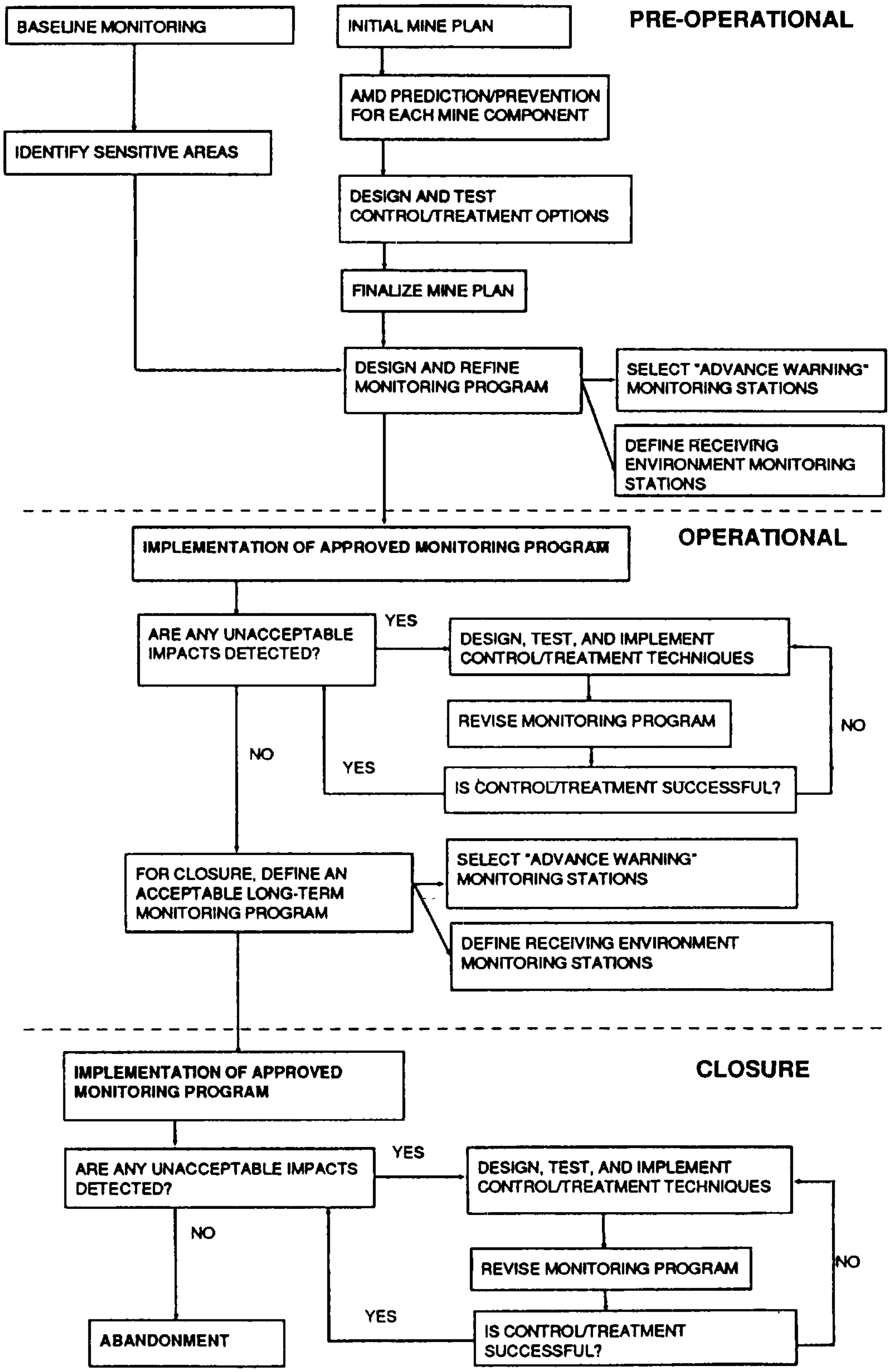
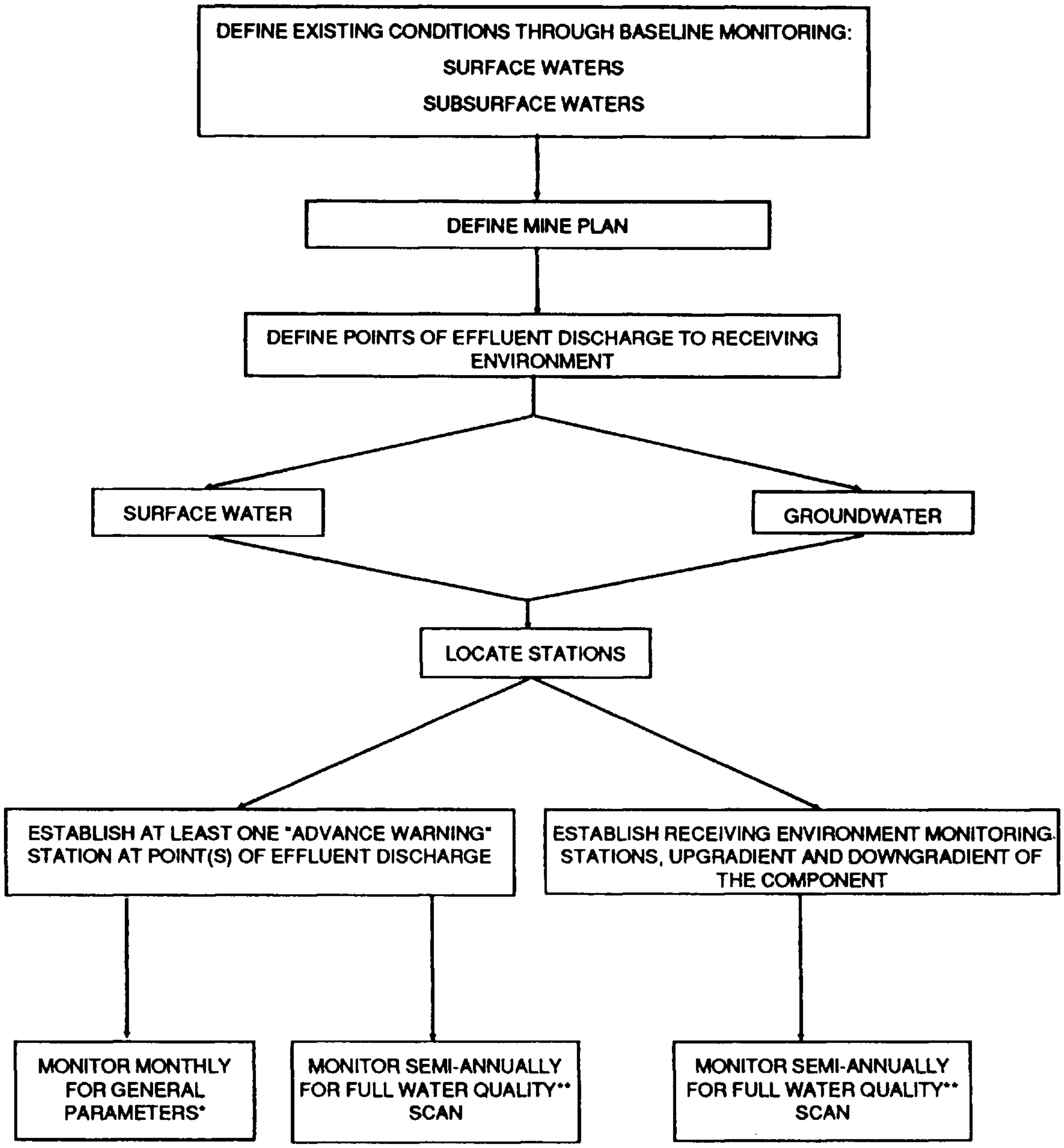


FIGURE 11.1-1
A RECOMMENDED APPROACH TO ENVIRONMENTAL MONITORING



* GENERAL PARAMETERS pH, SULPHATE, ELECTRICAL CONDUCTANCE ALKALINITY, ACIDITY, IRON

** FULL WATER QUALITY SCAN GENERAL PARAMETERS AND MAJOR CATIONS, ANIONS, NUTRIENTS, METALS

FIGURE 11.1-2
DESIGN AND REFINEMENT OF AN INTEGRATED ON-SITE/OFF-SITE MONITORING PROGRAM (for Operation and Closure)

A minimum of one surface water and/or groundwater monitoring station should be selected at a defined discharge point from each component to be an "advance warning" station in order to provide early warning of potential failure of acid prevention/control/treatment techniques. The advance warning stations are located at a point of direct discharge into the receiving environment from the mine component. Each mine component usually has at least one discharge point that can be selected as an advance warning station. These stations should be monitored at least monthly for: pH, sulphate, alkalinity, acidity, iron, and electrical conductance (Figure 11.1-2). A significant decrease in pH or alkalinity and an increase in sulphate may be an indication of the onset of acid drainage. However, extrsme care must be used in separating site-specific trends in water quality, such as seasonal variations in pH, from the onset of acid drainage.

Monitoring stations in the receiving environment in the vicinity of each mine component can be established within, upgradient of, and downgradient of the component for surface and groundwater flows. Downgradient stations, as defined by the movement of surface water or groundwater from the component, should be placed at various distances from the mine component in the receiving surface water and groundwater. Upgradient stations provide data for comparison with downgradient stations to determine the degree and spatial extent of impacts due to each component.

All stations should be monitored at least semi-annually for a full set of water quality analyses: pH, sulphate, alkalinity, acidity, iron, electrical conductance, major cations and anions, nutrients and a suite of metals. Stations monitoring may also be used for monitoring water flow rates.

Biological monitoring is not considered to be as reliable, as rapid, or as consistent as water-quality measurements and visual observations for the detection of acid drainage. Consequently, biological monitoring is not emphasized here for detecting acid drainage, although certain mines such as those located near important fisheries may be required to monitor productivity, species diversification, or metal levels in fish tissue. Nevertheless, an annual biological survey of the minesite and surrounding off-site region is recommended as contingency monitoring to check for changes in vegetation or fisheries which may indicate the migration of acid drainage not detected by an established monitoring network. As well, a visual inspection to identify changes in colour due to iron staining in seeps and streams in the area is recommended. This situation, for example, could arise during a first flush event where acid products are released between sampling periods of the monitoring stations.

The monitoring program implemented during the operational phase of the mine (Figure 11.1-1) should detect changes in environmental conditions at any station. If these changes are significant, more frequent monitoring should be performed at that station and at other stations to confirm the presence and spatial extent of the change. If an adverse impact is determined, alternative control or treatment techniques should be designed, tested, and implemented. The monitoring program should be revised to monitor the success of the new techniques.

If no unacceptable impacts exist towards the end of the mine life, a long-term monitoring program for closure would be defined (Figure 11.1-2). Each of the steps in Figure 11.1-2 should be performed because operational conditions at a mine ("baseline" for closure) will be different than pre-operational conditions.

The long-term monitoring program implemented during and after closure would decrease in frequency of sampling as time-from-closure increases. If significant changes in environmental conditions are detected at any station, additional monitoring should be performed at that station and at other stations to confirm the presence and spatial extent of the change. If the adverse impact is confirmed, alternative control or treatment techniques must be designed, tested, and implemented. The monitoring program must then be revised to monitor the success of the new techniques. If no unacceptable impacts are detected over an acceptably long period of time the site can proceed with abandonment.

11.2 SPECIFIC MONITORING PROGRAMS FOR EACH MINE COMPONENT

Each mine component will require monitoring for the effectiveness of treatment and control technologies and for the detection at the earliest time of failures of these systems. Therefore, the monitoring stations should include at least one advance warning station at a direct discharge point in addition to other stations located at different distances from the mine component in the receiving environment. As discussed above, the advance warning station should be monitored more frequently than other stations (Figure 11.1-2).

The objective of the advance warning stations is to detect any significant change from the background quality that indicates the on-set of acidic drainage from the mine component. Since the stations are monitored monthly for various parameters including pH, sulphate, alkalinity and acidity, trends can be established for background conditions.

The sampling frequency of the advance warning stations or other stations may be revised to incorporate a special sampling frequency. For example, at some sites there may be dry spells followed by heavy rains, therefore sampling of the first flushing of the waste rock stockpile after a storm event can be incorporated into the monitoring program.

Emphasis of the receiving environment monitoring program should be oriented directly downgradient from the minesite as defined by the direction of the surface and ground water flow from the site. Receiving environment monitoring defines existing conditions and identifies changes in the conditions during and after mining. Since the advance warning stations are the nearest to the area of impact and are monitored at frequent intervals to detect a failure in the treatment/control techniques, the receiving environment monitoring stations are to determine the longer term trends in water quality over a wider area.

A proposed monitoring program for each mine component is described in the following subsections. A summary of the surface and ground water flows around and through each component and how they are potentially impacted during and following mining operations is outlined. Some of the geotechnical aspects of each component related to geotechnical monitoring are also described. The minimum recommended monitoring program for each component is given along with a variety of options for monitoring each component.

11.2.1 Environmental Monitoring of Open Pits

11.2.1.1 INTRODUCTION

Potential sources of water entering a pit are precipitation, surface-water drainage, and groundwater discharge. A pit is open to the atmosphere so that variations in precipitation affect the daily volume of water in the pit, particularly if precipitation represents a major total of pit flow. Surface-water drainage is frequently diverted away from the pit perimeter in order to lower pumping costs. Groundwater frequently moves towards a pit because an excavation often represents a depression in the water table. Piezometers or monitor wells may be installed around the pit perimeter to intercept groundwater before it enters the pit if water control or geotechnical stability is a concern. All of the water entering a pit is usually combined and directed to sumps where it is pumped from the pit. The resulting surface flow downgradient of the pit may be directed into a holding pond for monitoring and any required treatment.

From a geotechnical viewpoint, a major concern is the stability of the pit walls. Instability represents a danger to the workforce and a delay in mining. Because water decreases friction through increase in pore pressures, groundwater can be a major cause of instability in a pit and may be controlled by interceptor wells or underground galleries at the pit perimeter.

11.2.1.2 MINIMUM RECOMMENDED MONITORING PROGRAM

During operation and at closure the location for the advance warning station in the vicinity of the open pit should be the discharge from the sump and/or the retention pond. The advance warning station is monitored once every month throughout operation and closure.

In addition, the seeps from the pit walls should be sampled once every six months to monitor the exposure of new zones over time and to determine the contribution of each wall to the acid rock drainage from the pit. Therefore, it may be possible to separate non-acidic and acidic flows from the pit walls to reduce the amount of impacted water.

Since there is no groundwater flow from the pit during operation no advance warning groundwater stations are necessary. After closure, discharge of mine water to the surface is terminated. This will allow the pit to flood and the water level in the pit will recover towards the pre-operational water table. As this occurs, the pit will become an integrated part of the groundwater flow system with groundwater flowing into, through, and out of the flooded pit. The surface water within the pit or flowing out of the pit can be established as the advanced warning station that will be monitored monthly at the closure phase.

Receiving environment surface monitoring stations should be located on surface watercourses that receive direct discharge from the sump and/or retention ponds holding pit water and at selected stream junctions downstream of the minesite. A minimum recommendad sampling program would be one sampling location in each surface water flow directly receiving discharge from the sump and/or retention pond. The minimum monitoring frequency for the surface water locations should be once every six months with the option for increased frequency depending on results of the advance warning station.

Groundwater monitoring at closure for the receiving environment is dependant on the hydraulic conductivity of the subsurface strata. If there are aquifers near the pit with high permeability, a groundwater monitoring network should be designed that is downgradient and hydraulically connected to the aquifers. The groundwater in an area that has a low permeability moves slower and monitoring stations located closer to the pit should be able to detect any changes. A minimum recommended sampling program would be one groundwater receiving environment sampling location in each aquifer in the area. The monitoring frequency for the groundwater locations should be once every six months.

11.2.1.3 MONITORING OPTIONS

During the operational phase of an open-pit mine, surface water is usually diverted away from the perimeter. As a result, any impacts on pre-existing surface drainage will be primarily a consequence of diversion. Monitoring of the diversion at established stations is an option but would be considered only if there was a concern with the ditch bank and bed material.

Precipitation may represent a major portion of pit water in wet climates. Precipitation and evaporation may be monitored at weather stations in or around the perimeter of the pit.

A pit often represents a depression in the local water table, resulting in groundwater flow towards and into the mine from the walls and bottom. This movement can be defined and monitored with piezometers and monitor wells installed around the pit perimeter. The flow rate and quality of groundwater can be monitored on pit walls, surface water quality can be monitored at the collector ditches within the pit, and at the sump and at the retention pond outside the pit.

11.2.2 Environmental Monitoring of Underground Workings

11.2.2.1 INTRODUCTION

The primary source of water in underground workings is groundwater because the workings often represent a depression in the local groundwater regime. A secondary source of water may be surface water directed into a shaft, adit, or decline intentionally for drilling water or unintentionally. Unlike an open pit (Section 11.2.1), water flow in underground workings does not usually respond as rapidly or as extremely as open pits because of the buffering process of infiltration to the groundwater system. Minewater flows under gravity to an adit or is pumped from the workings, resulting in a surface-water flow that may be directed to a retention pond.

Geotechnical concerns in underground workings focus on wall and roof stability to minimize dangers to the workforce and delays in mining. Additionally, any collapses in the workings may lead to increases in groundwater flow due to permeability enhancements and to subsidence on upper levels and tho land surface.

11.2.2.2 MINIMUM RECOMMENDED MONITORING PROGRAM

During operation and at closure the location for the advance warning station in the vicinity of the underground workings should be the discharge from the sump and/or the retention pond. The advance warning station should be monitored once every month.

In addition, the seeps from the walls could be sampled once every six months to monitor the exposure of zones over time and to determine the contribution of each wall or zone to the acid rock drainage from the workings. Therefore, it may be possible to separate the non-acidic and acidic flows to reduce the amount of impacted water.

After closure, discharge of mine water to the surface should be terminated. This will allow the workings to flood and the groundwater levels to recover towards pre-operational levels. As this occurs, the workings will become an integrated part of the groundwater flow system with groundwater flowing into, through, and out of the mine. The advance warning station would remain the same as during operation; the discharge point from the sump and/or retention pond.

Receiving environment surface monitoring stations should be located on surface watercourses that receive direct discharge from the underground workings and at selected stream junctions downstream of the minesite. A minimum recommended sampling program would be one sampling location in each surface water flow directly receiving discharge from the sumps and/or retention ponds. The minimum monitoring frequency for the surface water locations should be once every six months with the option for increased frequency depending on the water quality of the advance warning station.

Groundwater monitoring at closure for the receiving environment is dependant on the hydraulic conductivity of the subsurface strata. If there are aquifers near the underground workings with high permeability, a groundwater monitoring network should be designed that is downgradient and hydraulically connected to the aquifers. The groundwater in an area that has a low permeability moves slower and monitoring closer to the mine component should be able te detect any changes. A minimum recommended sampling program would be one groundwater receiving environment sampling location in each aquifer in the minesite area. The monitoring frequency for the groundwater locations should be once every six months.

11.2.2.3 MONITORING OPTIONS

During the operational phase of underground workings, surface water is usually diverted away from any shafts, declines or adits. As a result, any impacts on pre-existing surface drainage will be primarily a consequence of diversion. Monitoring of the diversion at established stations will indicate the extent of the impacts.

Underground workings often represent a local depression in hydraulio head, resulting in groundwater flow towards and into the mine. This de-watering of the groundwater system can be defined and monitored with piezometers and monitor wells installed from the land surface or frem the workings. Additionally, the flow rate and quality of groundwater can be monitored at stations in the workings, in the collector ditches within the workings or at any sumps and retention ponds outside the workings.

11.2.3 Environmental Monitoring Of Waste-rock Dumps, Ore Stockpiles, and Heap-leach Sites

11.2.3.1 INTRODUCTION

Dumps, stockpiles, or heap-leach sites are usually exposed to climatic events so that precipitation represents a primary source of water moving through the rock. The resulting infiltration moves downward under gravity through the rock to the base of the pile. At the base, the infiltration may (1) mix with upwelling groundwater and flow from the basal perimeter of the pile, (2) completely enter the underlying groundwater flow system if hydraulic conductivity is sufficiently high to accept all infiltration and hydraulic gradients have a downward component, and (3) partially enter the groundwater system if hydraulic conductivity is restricted and partially exit at the basal perimeter of the pile. Heap-leach sites may be constructed with low-permeability pads so that negligible amounts of infiltration reach the underlying groundwater system during the active life of the site.

Geotechnical monitoring of dumps, stockpiles, and heap-leach sites address the physical integrity of the rock after placement. Consolidation/settlement and slope stability could cause changes in the hydraulic conditions within the structure and could cause migration of rock from the site through physical movement such as slumping and toe collapse.

11.2.3.2 MINIMUM RECOMMENDED MONITORING PROGRAM

During operation and at closure (if the piles are not being removed), the location of the advance warning stations should be at the discharge point from a retention pond or drainage ditches located at the base of the piles. If there is primarily groundwater drainage due to the hydraulic conductivity in the area, there should be additional advance warning stations requiring the installation of a groundwater well network. Both surface and groundwater advance warning stations should be monitored once a month.

Receiving environment surface monitoring stations should be located on surface watercourses that receive direct discharge from the retention pond or ditches around the pile and at selected stream junctions downstream of the minesite as well as one upgradient surface water station. A minimum recommended sampling program would be one upgradient and downgradient sampling location in each surface water flow directly receiving discharge from the pile. The minimum monitoring frequency for the surface water locations should be once every six months with the option for increased frequency depending on the water quality at the advance warning station.

Groundwater monitoring at closure for the receiving environment is dependant on the hydraulic conductivity of the subsurface strata. If there are aquifers near the pile with high permeability, a groundwater monitoring network should be designed that is downgradient and hydraulically connected to the aquifers in addition to one upgradient groundwater station. The groundwater in an area that has a low permeability moves slower and monitoring locations closer to the mine component should be able to detect any changes. A minimum recommended sempling program would be one upgradient and one downgradient groundwater sampling locations in each aquifer in the receiving environment. The monitoring frequency for the groundwater locations should be once every six months.

11.2.3.3 MONITORING OPTIONS

Hydraulic monitoring of dumps, stockpiles, and heap-leach sites may include periodic measurements of precipitation at exposed surfaces and with periodic measurements of rates and quality of infiltration into the rock. Within a pile, monitoring would indicate the preferred water pathways and the variation in quality during flow, although monitoring of water in unsaturated, coarse-grained material is difficult and not always feasible. If there is a water table within the pile a piezometer could be installed to monitor the water quality. Additionally, subsurface stations consisting of piezometers or monitor wells may be installed around the perimeter to monitor the direction, flow rate, and quality of groundwater in the area.

Monitoring of the temperature gradients may provide early indications of changes within the pile and can be accomplished using temperature probes within the piles. This method may indicate the initiation of acid generation in localized areas.

11.2.4 Environmental Monitoring of Tailings Impoundments

11.2.4.1 INTRODUCTION

Two primary sources of water entering a tailings impoundment are the mill effluent discharge and precipitation. Water remaining above the tailings surface will flow to low-lying areas and form ponds. From a groundwater perspective, high water levels in an impoundment lead to downward movement through the tailings pile with a lateral component of flow towards the perimater. Groundwater may leave an impoundment through the base or the retaining walls and enter the local groundwater flow system. This seepage may then enter deeper flow systems or discharge locally to the surface.

11.2.4.2 MINIMUM RECOMMENDED MONITORING PROGRAM

During operation and at closure the location of the advance warning station should be at the direct discharge points from the impoundment. Direct discharge points can include seepage locations through dams and embankments to monitor the groundwater quality, and discharge from the spillways to monitor the surface water quality. These discharge locations should be monitored every month.

When the tailings become consolidated after closure additional advance warning stations can be located within the tailings impoundment by installing piezometers in the tailings to monitor the quality of the groundwater pore water.

Receiving environment surface monitoring stations should be located on surface watercourses that receive direct discharge from the tailings impoundment and at selected stream junctions downstream of the impoundment. A minimum recommended sampling program would be two sampling locations in each surface water course directly receiving discharge from the tailings impoundmant, one upgradient and one downgradient. The minimum monitoring frequency for the surface water locations should be once every six months with the option for increased frequency depending on water quality at the advance warning station.

Groundwater monitoring at closure for the receiving environment is dependant on the hydraulic conductivity of the subsurface strata. If there are aquifers near the tailings impoundment with high permeability, a groundwater monitoring network should be designed that is downgradient and hydraulically connected to the aquifers. The groundwater in an area that has a low permeability moves slower, monitoring closer to the impoundment should be able to detect any changes. A minimum recommended sampling program would be one upgradient groundwater station for each hydraulically connected strata impacted by the tailings impoundment and one groundwater sampling locations in each aquifer in the minesite area. The monitoring frequency for the groundwater locations should be once every six months.

11.2.4.3 MONITORING OPTIONS

Hydraulic monitoring within an active impoundment is not always possible due to the unconsolidated nature of fresh tailings, sometimes limiting access only to the perimeter dams and retaining walls. As a result, measurements of mill effluent rate and quality can be made at the mill. Measurements of local precipitation and evaporation can be made at the perimeter. The movement of surface water within the impoundment can be visually defined and the movement of groundwater can be generally assessed at the perimeter with piezemeters.

Geotechnical monitoring of an impoundment at established stations address the physical integrity of a impoundment during and after operation. Dam instability, for example, may lead to the migration of tailings solids from an impoundment to a downstream environment. Tailings settlement may affect the rate and direction of water movement, which is a primary pathway for the interaction of tailings with the surrounding environment.

11.2.5 Environmental Monitoring of Quarries

11.2.5.1 INTRODUCTION

Like open pits (Section 11.2.1), potential sources of water entering a quarry are precipitation, surface-water drainage, and groundwater discharge. A quarry is open to the atmosphere and surface-water drainage is not usually diverted around the site so that variable flow rates through the quarry can be expected. Groundwater frequently moves towards a pit because an excavation often represents a depression in the water table; however, a shallow quarry may lower the water table below the base so that groundwater inflow becomes minimal. All of the water entering a quarry usually combines and is either pumped out or flows under gravity from the site. The resulting surface flow downgradient of the quarry may be directed into a holding pond for monitoring and any required treatment.

The prevention of acid generation from a quarry should primarily be the assessment of its acid generation potential. The use of a potentially acid generating quarry is unlikely, therefore, a minimum monitoring program is only to confirm that acid generation is not occurring.

From a geotechnical viewpoint, the major concern is the stability of the quarry walls while equipment and workers are in the area. Shallow quarries may be relatively stable without any stability engineering.

11.2.5.2 MINIMUM RECOMMENDED MONITORING PROGRAM

The advance warning station can be located at the retention pond or from the surface discharge flow from the quarry and is monitored once a month. No groundwater advance warning stations are necessary as not all quarries will have a groundwater component. Receiving environment surface monitoring stations are optional (Section 11.2.5.3).

11.2.5.3 MONITORING OPTIONS

The water derived from a quarry may be monitored for flow rate and quality. If unacceptable quality is anticipated, this water may be directed to a holding pond for analysis and treatment.

The groundwater monitoring component for receiving environment monitoring is dependant on the hydraulic conductivity of the subsurface strata. If there are aquifers near the quarry with high permeability, a groundwater monitoring network should be designed that is downgradient and hydraulically connected to the aquifers. The groundwater in an area that has a low permeability moves slower and monitoring closer to the quarry should be able to detect any changes.

Receiving environment surface monitoring stations could be located on surface watercourses that receive direct discharge from the quarry and at selected stream junctions downstream of the quarry. An optional sampling program could be one sampling location in each surface water flow directly receiving discharge from the quarry.

11.2.6 Environmental Monitoring of Haul Roads

11.2.6.1 INTRODUCTION

For road construction, stability, and maintenance, rock is often taken from mine quarries and mine waste rock. This rock is crushed to an appropriate size and is further crushed by road vehicles. As a result, these roads can be thought of as small-scale waste-rock dumps (Section 11.2.3), but their geochemical reactivity is higher because of the finer grain size and continual grinding. Precipitation onto the roads passes through the rock and either enters the underlying groundwater system or moves as overland flow to low-lying areas. Since the prevention of acid rock drainage from a haul road is in the assessment of acid generation prior to use of the material, the construction of an acid generating haul road is unlikely. But, due to use of marginal acid generating material or material used without knowledge of its acid generating capability a minimum monitoring program is proposed.

11.2.6.2 MINIMUM RECOMMENDED MONITORING PROGRAM

Since monitoring of acid generation from haul roads may be very difficult the minimum recommended program would be a visual inspection every six months of the area adjacent to the haul roads to check for discoloration of local materials. Stations may be located in sediment traps if they have been constructed and would be monitored every month.

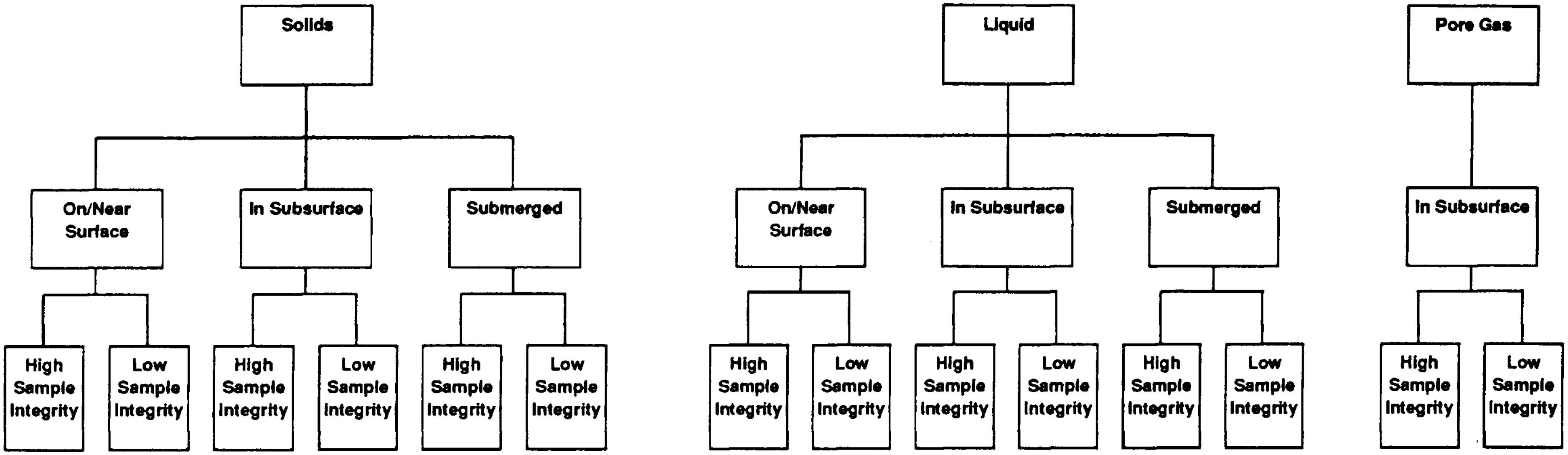
11.2.6.3 MONITORING OPTIONS

Monitoring of overland flow can be accomplished by defining drainage patterns from a road and establishing monitoring stations (Figures 11-1 and 11-2) along flowpaths. Groundwater can be monitored by installing shallow monitor wells perpendicular to the road length.

11.3 FIELD METHODS FOR SOLID, LIQUID AND POREGAS SAMPLING

There are numerous methods available for on-site and receiving environment monitoring for acid rock drainage impacts. Figure 11.3-1 provides a flowchart to select the sampling method required for various types of samples, either on or near surface, in the subsurface or submerged. The methods are further separated to high and low sample integrity. The methods are described along with its advantages and disadvantages in Field Sampling Manual for Reactive Sulphide Tailings (Canect Environmental Control Technology, 1989).

FIGURE 11.3-1
Flow Chart for Selection of Sampling Method



Solids						Liquid						Pore Gas	
On/Near Surface		In Subsurface		Submerged		On/Near Surface		In Subsurface		Submerged		In Subsurface	
High Sample Integrity	Low Sample Integrity	High Sample Integrity	Low Sample Integrity	High Sample Integrity	Low Sample Integrity	High Sample Integrity	Low Sample Integrity	High Sample Integrity	Low Sample Integrity	High Sample Integrity	Low Sample Integrity	High Sample Integrity	Low Sample Integrity
Hand-driven core (H)[a] Sampling boxes(H)	Trenching (L)[b] Shovel/bucket (H) Hand auger (H) Small hand-operated samplers (H)	Split spoon (L) Shelby tube (L) Piston Sampler (L) Vibratory core (L) University of Sherbrooke block sampler (L) Core-freezer sampler (L)	Auger flights (P)[c] Rotary drill (P) Hammer drive (P) Cable tool (P)	Gravity coring (L) Sampling Box for submerged samples (H)	Grab sampling(L) Inverting Bottom Sampler (H) Small hand-operated samplers (H)	Surface-water grab		Monitor well Piezometer Suction lysimeter (teflon) Peristaltic pump Piston pump Bladder pump Waterra pump Syringe sampler Bailer, double valve	In-filling of stem Advance intake beyond stem Centrifugation Pressurized consolidation Porewater displacement Suction lysimeter (ceramic) Wash-boring/ jetting Centrifugal pump Gas-driven pump Bailer, single valve	Seepage meters		Closed circuit port Single port with syringe sampler Multiple port with syringe sampler	Withdrawn from drill stem Withdrawn from port extended beyond stem Single port with autoanalyzer Multiple ports with autoanalyzer

a H - handheld sampler
b L - lightpowered
c P - heavy powered

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APPENDIX 4.4

DESCRIPTION OF STATIC TESTS

APPENDIX 4.4

4.4 DESCRIPTION OF STATIC TEST ANALYSES

4.4.1 Total sulphur

Objectives

- o To determine the total sulphur content of a sample; and
- o To allow the calculation of maximum potential acidity in a sample.

Description of test

Total sulphur is usually determined using a Leco sulphur analyzer. The sample is heated to approximately 1600°C with a stream of oxygen passing through the sample. Sulphur dioxide is released from the sample and collected in a solution, which is then titrated to determine the total sulphur. Total sulphur is usually reported as the percentage of sulphur relative to the entire sample (%S).

Interpretation

Total sulphur (as %S) is converted to maximum potential acidity in units of kilograms of $CaCO_3$ equivalent/tonnes of sample (or t of $CaCO_3$ equivalent/1000 t of sample) through multiplication by 31.25. This conversion factor is derived as follows. Firstly, it is assumed that the pyrite is completely oxidised by oxygen and water to sulphate and solid $Fe(OH)_3$.

$FeS_2 + 7/2H_2O + 15/4O_2 \text{-->} 4H^+ + Fe(OH)_3 + 2SO_4^{2-}$

Then, it is assumed that hydrogen ions produced in the reaction are incompletely neutralized by $CaCO_3$ to a pH not greater than 6.

$2H^+ + CaCO_3 \text{-->} Ca^{2+} + H_2CO_3$

Based on this reaction pair, stochiometrically the acidity produced by 1 mole of sulphur is neutralized by 1 mole of $CaCO_3$. One gram of sulphur in 100 g of material (1% S) is equivalent to 0.03125 moles of sulphur which would be neutralized by 0.03125 moles or 3.125% $CaCO_3$. This concentration is conventionally expressed as 31.25 kg $CaCO_3$/tonne of material. Thus, the conversion factor is theoretical and is based on geochemical assumptions depending on the acid-generating conditions. Realistically, the conversion factor could be significantly greater than or less than 31.25.

Advantages

- theoretical maximum amount of acidity that could be produced is calculated from total sulphur;
- the method is widely used and proven; and
- the measurement provides confirmation of cumulative quantities from other mineral specific tests for sulphur (Appendix 4.4, Seotion 4.4-2, Appendix 4.5, Section 4.5.1).

Disadvantages

- acid-generating and non-acid-generating sulphur minerals are not distinguished; and
- rates of acid generation are not predicted.

4.4.2 Sulphur species

Objectives

- to determine the acid-leachable sulphate, sulphide and non-leachable sulphur content in a sample; and
- to allow the calculation of potential acidity in a sample based on sulphide content.

Description of Test

The sample is split and tests are conducted on subsamples. The first subsample is reacted with heated hydrochloric acid (HCl), then precipitated with barium chloride and the barium sulphate precipitate is weighted and calculated for sulphur. This HCl-extractable sulphur represents acid-leachable sulphate. The second subsample is reacted with nitric acid (HNO_3), then analyzed for sulphur using the same method as above. The difference between the HCl leach sulphur and the HNO_3 leach sulphur represents leachable sulphide.

The sulphur remaining in the HNO_3 - leached subsample represents non-leachable sulphur which is stable during the time period of the leach. This subsample should be analyzed for total sulphur to confirm that all of the original sulphur is accounted for. The non-leachable sulphur may represent potentially acid-generating sulphur such as elemental sulphur and non-acid-generating sulphur such as barite (barium sulphate). An analysis for barium would indicate potential barite content.

Interpretation

The sulphide content as %S is converted to potential acidity through multiplication with 31.25 (see Appendix 4.4, Section 4.4.1).

Advantages

- potential acidity represents only rapidly oxidized sulphide and does not includes non-acid generating forms of sulphur.

Disadvantages

- highly reactive sulphide may be leached by the heated HCl, therefore, the HCl extractable "sulphate" may actually be a combination of sulphate and highly reactive sulphide;
- sulphur which is initially slow to oxidize during the acid leaching, but which is capable of generating significant acidity under reasonable environmental conditions, is not detected.
- the isolated extraction of sulphate and sulphide has not been conclusively demonstrated and therefore the method requires further research and development; and
- rates of acid generation are not predicted.

4.4.3 Reactive sulphur

Objective

- to determine the amount of reactive, acid-generating sulphur contained in a sample.

Description of Test

The sample is first quickly prewashed with water to remove soluble sulphate. The washed material is separated into two subsamples and both subsamples are mixed with a known amount of hydrogen peroxides (40%) in water. One subsample is left for one day and the other is left for one week. After the required length of time the water is separated from the material and is analyzed for total sulphur and sulphate content.

Interpretation

The amount of reactive sulphur can be determined from the amount of sulphate produced from the test. This reactive sulphur is converted to potential acidity through multiplication by 31.25 (Appendix 4.4, Section 4.4.1).

Advantages

- potential acidity based on the highly reactive sulphur can be determined.

Disadvantages

- method is in development stage and has not been reliably verified;
- the potential exists for "non-reactive" sulphide to generate acidity over extended periods of time; and
- rates of acid generation are not predicted.

4.4.4 Gross Neutralization Potential

Objectives

- to determine the maximum neutralization potential of a sample.

Description of Test

The total amount of neutralizing minerals including carbonates and hydroxides present in the material is determined by treating a sample with a known excess of standardized hydrochloric acid. The sample and acid mixture is heated to ensure the reaction between the acid and neutralizing minerals is complete. The amount of unconsumed acid is then determined by titrating with standardized sodium hydroxide to pH 7.

Interpretation

Neutralization potential is calculated by converting the amount of base to a calcium carbonate ($CaCO_3$) equivalent, commonly in units of kg/t of sample, t/1000 t, or % carbon dioxide (CO_2). This expression of neutralization potential as $CaCO_3$ is deceiving in that most natural neutralizing minerals are not capable of neutralizing pH above a value of 6. Furthermore, there is some uncertainty in the hydroxide titration in that some minerals will re-precipitate, but at varying rates, which affects the amount of hydroxide needed to reach pH 7.

Advantages

- the maximum amount of neutralization potential available in a sample may be determined; and

- the method is widely used and has been proven to be reliable as a gross measure of total neutralization.

Disadvantages

- the method does not indicate the pH to which the sample can neutralize during extended contact with acidic water; and
- rates and extent of neutralization are not predicted.

4.4.5 Carbonate Analysis

Objective

- to determine the amount of strongly neutralizing, carbonate minerals in a sample.

Description of Test

Carbonate is determined by digesting the sample in an enclosed chamber with acid and heat so that carbon dioxide (CO_2) gas is evolved. The CO_2 gas is absorbed into a basic solution and is measured using Leco analyzer.

Interpretation

The amount of CO_2 gas evolved is used to calculate the amount of carbonate present in the material. The result is converted to calcium carbonate equivalent.

Advantages

- only carbonate minerals which are generally capable of neutralizing pH to 5-9 are determined; and
- the method is widely used but is not widely used for AMD prediction and proven to be reliable.

Disadvantages

- detection limit of 2 t $CaCO_3$/1000 t of sample is not low enough for predicting relatively short-term neutralization of pH, although it is sufficient for acid-base accounting purposes; and
- cannot be used when the pyrrhotite content is greater than 10% due to the production of SO_2 which interferes with CO_2 in the Leco analyzer.
- rates and extent of neutralization are not predicted.

4.4.6 Low detection limit carbonate analysis

Objective

- to determine the amount of strongly neutralizing, carbonate minerals in a sample.

Description of Test

The sample is placed in a glass syringe and the sample is purged with nitrogen. A strong acid is drawn into the syringe and allowed to react with the sample for several hours. The gas in the headspace with the evolved CO_2 is injected into a gas chromatograph for analysis and quantification.

Interpretation

The amount of CO_2 gas evolved is used to calculate the amount of carbonate present in the sample. The result is converted to calcium carbonate equivalent.

Advantages

- low detection limit allows prediction for short-term neutralization of pH; and

- only the carbonate minerals which are generally capable of neutralizing pH to 5-9 are determined.

Disadvantages

- rates and extent of neutralization are not predicted; and
- the method is not used routinely.

4.4.7 Paste pH

Objectives

- to determine the pH that the sample can immediately produce upon contact with water;

- to measure the capacity for immediate dissolution of calcium carbonate in a sample; and

- to indicate whether significant acid generation occurred prior to or during paste-pH analysis.

Description of Test

Paste pH is measured by a pH/reference electrode assembly and a pH meter. The electrode assembly is placed into the paste formed by mixing water and powdered rock in a specific ratio and the pH is read from the meter.

Interpretation

A paste pH greater than the pH of the mixing water indicates immediate neutralization has occurred and a pH above 7 suggests either the presence of reactive calcite or the contamination by drilling fluids. A paste pH of less than 4.0 indicates that the material contains readily available acidity from prior acid generation and the material is toxic to most plants.

Advantages

- the pH is simple and easy to obtain; and
- the pH indicates whether the sample contains readily available acidity or alkalinity.

Disadvantages

- the duration of the acidic or alkaline condition of the sample cannot be predicted; and
- rates of acid or alkaline generation are not predicted.

APPENDIX 4.5

SAMPLE CHARACTERIZATION AND DESCRIPTION OF KINETIC TESTS

APPENDIX 4.5

4.5.1 SAMPLE CHARACTERIZATION

4.5.1.1 Particle size

Objectives

- to determine the size distribution of particles in the sample; and
- to allow an approximate calculation of surface area.

Description of Test

Coarse material should be dry sieved using stainless steel wire mesh; however, wit sieving may be necessary if the material contains fines. The sieves used should reflect the overall texture of the sample with screens selected from the scheme listed in Table 4.5.1-1.

Coarser screens may be added at the top end, though material finer than 0.03 mm (30 um) cannot be sieved and should be determined using a different method such as a pipette or hydrometer method. Very coarse samples must be very large in order to be representative.

After sieving each fraction is weighed.

Calculation of Surface Area

The data may be used to determine mean diameter of the material and total surface area. Since the size distribution is assumed to be geometric (Table 4.5.1-1) diameters of fractions should be expressed as the log diameter (conventionally called phi). This assures that errors in measurement of the coarser fractions do not overwhelm the finer fractions.

TABLE 4.5.1-1
SIEVE SIZES

OPENING (mm)	ASTM MESH	DIAMETER IN PHI[a]
4	5	-2
2	10	-1
1	18	0
0.5	35	1
0.25	60	2
0.125	120	3
0.0625	230	4
0.03125	325	5

[a] Phi = $-\log_2(\text{mm})$

The total surface area is given by:

$$S = \frac{1}{g} \quad (3Mi/(di)/2)$$

Where di and Mi are the diameter and weight of material in fraction i respectively. The variable, di, is usually taken as the mid-point of the bounding sieves. The variable g, is the representative density of the solids, which can be defined through a density measurement. The solids density may also be estimated from mineralogy (Appendix 4.5, Section 4.5.1.2) and will vary from 2.7 g/cm3 (low density silicates) to 5/g/cm3 (pyrite).

<u>Advantages</u>

- o method is readily available and inexpensive.

<u>Disadvantages</u>

- o assumes that all particles are spheres;
- o does not yield exposed areas of sulphides and neutralizing minerals; and
- o additional research is required to determine reactive sulphide particle size.

4.5.1.2 Mineralogy

Objectives

- to determine the types and abundances of minerals in samples.

Description of Test

Mineralogy can be determined from hand specimens using a hand lens or binocular microscope. Identification of fine-grained mineral grains is carried out using polished mounts, thin sections or polished thin sections. The latter permit identification of transparent and opaque minerals simultaneously. Very fine grained minerals can be determined by x-ray diffraction (XRD) and scanning electron microscope (SEM).

Advantages

- provides data for later interpretation of static and kinetic tests.

Disadvantages

- relationship of mineralogy to rates of acid generation or neutralization may not be apparent unless post-test mineralogy is defined for comparison.

4.5.1.3 Total metals analysis

Objectives

- to determine metals present in samples so that depletion from the sample during testing can be evaluated.

Description of Tests

A variety of tests are available most of which involve wet chemical digestion followed by spectrographic analysis. Complete digestion of all minerals is achieved with hot mixed acids (hydrochloric-perchloric-nitric). Alternatively, a number of digestions of varying strengths are available which allow leaching of specific minerals. For example, aqua regia (3:1, hydrochloric-nitric acid) will not completely digest strongly resistant oxides and silicates. These selective extractions must be complemented by an understanding of sample mineralogy (Appendix 4.5, Section 4.5.1.2).

The most commonly used spectrographic methods are: inductively coupled plasma-emission spectroscopy which is a simultaneous multi-element method and atomic absorption spectroscopy which is a single-element method.

Advantages

- permits selection of potentially problematic species for chemical analysis during a test;
- allows changes in leachate composition to be related to mineralogy.

Disadvantages

- the amount of metal available for leaching may be overestimated due to metal content within sample particles that are unavailable for leaching; and
- digestion of geological materials requires further research to determine the digestion of specific metals in a complex matrix.

4.5.2 KINETIC TEST DESCRIPTIONS

4.5.2.1 B.C. confirmation tests

Objectives

- to confirm the results of static tests.

Description

The sample is placed in a flask with sufficient sulphuric acid to attain a pH of 2.5 and shaken for four hours with acid added as necessary to maintain a pH of 2.5 to 2.8. Active culture of Thiobacillus ferrooxidans is then added to the flask and the flask is weighed. The flask is then plugged with loose cotton and incubated at 35°C while continuously shaken. The pH and one dissolved metal is monitored for the first three days to ensure pH remains below 2.8 with distilled water added to the flask to maintain constant weight. Once a pH below 2.8 is ensured, monitoring is performed every second day until microbiological activity ceases, which is defined by no further change in pH and dissolved metal concentration. At this point, one-half of the original weight of sample is added to the flask and shaken for 24 hours. If pH rises above 3.5, the test is terminated. If pH is less then 3.5, additional sample is added at one-half the original weight and shaken for 24 hours. If pH is less than 3.5 or greater than 4.0, the test is terminated. Otherwise, the flask is shaken another 48 hours and pH is recorded.

Interpretation

The objective of the test is to determine if the bacteria can oxidize enough sulphides to sulphuric acid to satisfy its acid demands. If the bacteria can remain self sustaining then there is a strong possibility that acid mine drainage will occur. If there is insufficient acid production, the pH will approach the natural pH of the sample above pH 3.5, then the sample is interpreted as a non-acid producer. If the pH remains in the leaching range of below pH 3.5, then there is a strong possibility that the sample is an acid producer.

Advantages

- relatively simple to use.

Disadvantages

- the test apparently confuses or ignores the concepts of solid-liquid balance, buffering and neutralization, inorganic sulphide oxidation time, the optimum pH range for T. ferrooxidans activity, and microbiological toxicity in a closed environment;
- rates of acid generation or neutralization are not predicted;
- control techniques for acid generation cannot be tested; and
- methods that have been developed recently can indicate more reliably the confirmation of acid generation.

4.5.2.2 Soxhlet reactor

Objectives

- to confirm the results of static tests.

Description of Test

A standard Soxhlet reactor method is used with deionized water placed in the reservoir and the sample in the thimble. Tho water in the reservoir is heated and vaporized and passes into the condenser. The condensed liquid drips onto the sample in the thimble and then back into the reservoir. The leachate is analyzed after 64 to 192 hours. Variations include intermittent drying of the sample and operating the reactor at discrete intervals.

Interpretation of the Results

The water in the reservoir holds any products of acid generation. In order to confirm the results of static tests, the pH can be evaluated as follows.

pH: <3.0 = sample is strongly acid generating

3.0-5.0 = sample is acid generating and some pH neutralization may be occurring

>5.0 = sample is not significantly acid generating or neutralization of pH is overwhelming the acid generated.

Advantages

- results are available after a relatively short period of time;
- various control options can be tested to a limited extent; and
- the test is relatively simple to set up and operate.

Disadvantages

- results cannot be used to simulate natural conditions;
- method is used infrequently;
- the influence of bacteria on the rate of acid generation cannot be monitored; and
- boiling and refluxing of water may significantly change the geochemistry.
- method is still in the developmental stage;

4.5.2.3 Shake flasks

Objectives

- to determine the rate of and temporal variations in acid generation;
- to determine the temporal variation in overall water quality;
- to confirm the results of static tests; and
- to test some proposed control options.

Description of Test

The sample is placed in a flask and covered with excess water or nutrient solution. The water allows testing of effects of varying pH and water chemistry over time. Additionally, the flask can be inoculated for microbiological testing and temperature can be varied. The flask is shaken continuously and periodically sampled for water quality parameters including sulphate concentration. The sampling involves the removal of sufficient water for analysis from the flask. If the volume of the removed water is a significant portion of the total water the flask should be refilled, although this will complicate the interpretation of the test.

Interpretation of Results

The water within a flask represents a medium into which the products of acid generation accumulate through time. The monitoring of the accumulation process will indicate temporal trends in the chemical reactions. General guidelines for evaluating the test include pH, sulphate, acidity, and metals.

pH: <3.0 = sample is strongly acid generating

3.0-5.0 = sample is acid generating and some pH neutralization may be occurring;

>5.0 = sample is not significantly acid generating or neutralization of pH is overwhelming the acid generated.

Sulphate: Sulphate is a direct by-product of complete sulphide oxidation. As a result, sulphate is indicative of the rate of sulphide oxidation and acid generation if:

- all oxidized sulphur is released to the water;
- all sulphur in the water is fully oxidized to sulphate; and
- precipitation of gypsum or another sulphate mineral does not limit aqueous sulphate concentration.

An increase in sulphate concentrations from analysis to analysis indicates that sulphate is accumulating in the unflushed water, providing the above conditions are met. However, initial analyses may indicate only the dissolution of sulphate minerals or the release previously accumulated acid products. It is possible that sulphate may indicate a strong rate of acid generation although pH is neutral, due to the presence of carbonate minerals. Rates of acid ganeration and sulphide oxidation can be calculated on a weight basis (for example, mg SO_4/kg/week or mg $CaCO_3$ equivalent/kg/week), on a surface-area basis (for example mg SO_4/m^2/week) derived from surface-area- to-weight conversions (Appendix 4.5, Section 4.5.1.1), and on a cumulative basis (for example total mg SO_4/kg).

Acidity: Acidity is a cumulative measure of several aqueous species such as Fe^{2+}, $Fe(OH)^{2+}$, Al^{3+}, and HSO^{4-}. As a result, acidity can be used as a gross indicator of acid generation. A correlation between sulphate and acidity frequently exists unless neutralization is significant. Rates can be calculated on a weight basis (for example, mg $CaCO_3$ equivalent/kg/week), on a surface-area basis or on a cumulative basis.

Metals: Analyses of aqueous metals will indicate the rate of metal depletion from the sample by the comparison of metal release to the original content of the metal (Appendix 4.5, Section 4.5.1.3). The release of metal from sulphide minerals during oxidation is usually not a reliable indicator of the rate of acid generation due to secondary effects such as hydrolysis and precipitation of secondary minerals.

Advantages

- rates of acid generation and neutralization in a simulate submerged, high-energy environment can be determined;
- effects of different environmental factors such as pH and temperature can be tested during the same time interval;
- temporal variations in rates of acid generation and neutralization can be determined;
- various control options such as limestone addition can be tested;
- influence of bacteria on the rate of acid generation may be monitored; and
- the test is relatively simple to set up and operate.

Disadvantages

- test is not widely used in British Columbia so reliability is undefined;
- bacteria may not acclimatize to the sample;
- high water content in flask may inhibit acid production;
- a long time may be required to complete test to meet program objectives; and
- interpretation of data is complex.

4.5.2.4 Humidity Cells

Objectives

- to determine the rate of and temporal variation in acid generation;

- to determine the temporal variation in overall water quality;
- to confirm the results of static tests; and
- to test some proposed control options.

Description of Test

The samples are placed in enclosed chambers with an entrance port for connection to the continuous flow air supply and an exit port to ensure there would be no pressurization in the cell. Pre-filtered humidified air is supplied to the cell (Figure 4.5.2-1). Distilled water is periodically added, the sample is allowed to soak for about an hour, then the water is decanted. Experience has shown that a single wash may not be sufficient to remove all accumulated acid products so that a second wash may be conducted and composited with the first wash. Various modifications of the technique include inoculation with an active culture of Thiobacillus ferrooxidans, wet/dry cycles, full saturation or complete submergence of a sample, and humid air passing directly upwards through the sample.

Interpretation of Results

The chemical analysis of each periodic rinse will indicate the extent of acid generation, sulphide oxidation, neutralization and metal depletion since the previous rinse. If the time between rinses is relatively long, data must be evaluated to determine if the accumulation of acid products has slowed reaction rates. General guidelines for evaluating the test include pH, sulphate, acidity, and metals.

pH: < 3.0 = sample is strongly acid generating

3.0-5.0 = sample is acid generating and some pH neutralization may be occurring

> 5.0 = sample is not significantly acid generating or neutralization of pH is overwhelming the acid generated.

Sulphate: Sulphate is a direct by-product of complete sulphide oxidation. As a result, sulphate is indicative of the rate of sulphide oxidation and acid generation if:

- all oxidized sulphur is released to the water;
- all sulphur in the water is fully oxidized to sulphate; and
- precipitation of gypsum or another sulphate mineral does not limit aqueous sulphate concentration.

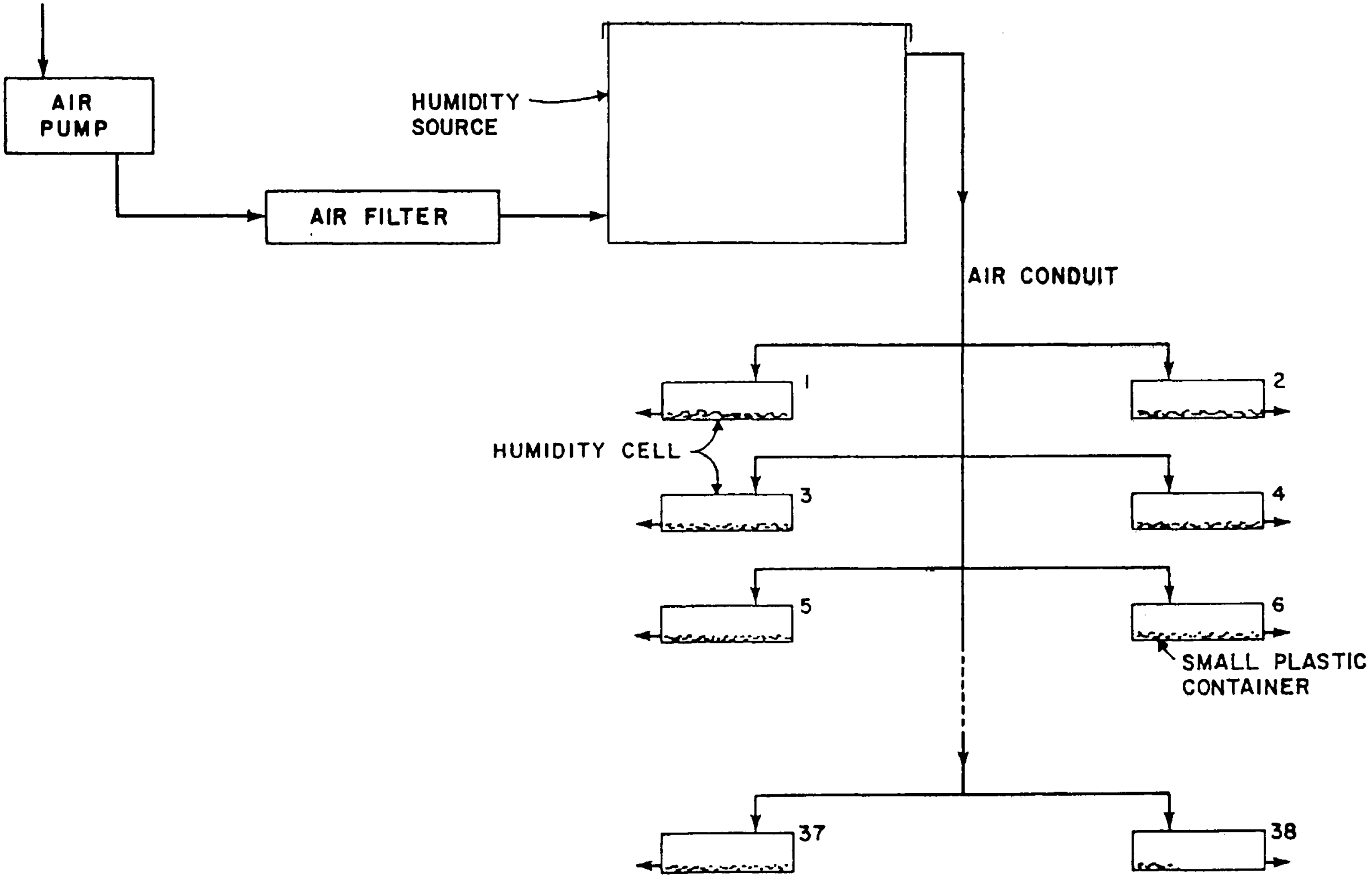

FIGURE 4.5.2-2
EXAMPLE OF A HUMIDITY CELL APPARATUS

An increase in sulphate concentrations from analysis to analysis indicates the rate of acid generation is accelerating, providing the above conditions are met. However, initial analyses may indicate only the dissolution of sulphate minerals or the release of previously accumulated acid products. It is possible that sulphate may indicate a strong rate of acid generation although pH is neutral, due to the presence of carbonate minerals. Rates of acid generation and sulphide oxidation can be calculated on a weight basis (for example, mg SO_4/kg/week or mg $CaCO_3$ equivalent/kg/week), on a surface-area basis (for example mg SO_4/m^2/week) derived from surface-area- to-weight conversions (Appendix 4.5, Section 4.5.1.1), and on a cumulative basis (for example total mg SO_4/kg).

Acidity: Acidity is a cumulative measure of several aqueous species such as Fe^{2+}, $Fe(OH)^{2+}$, Al^{3+}, and HSO^{4-}. As a result, acidity can be used as a gross indicator of acid generation. A correlation between sulphate and acidity frequently exists unless neutralization is significant. Rates can be calculated on a weight basis (for example, mg $CaCO_3$ equivalent/kg/week), on a surface-area basis or on a cumulative basis.

Metals: Analyses of aqueous metals will indicate the rate of metal depletion from the sample by the comparison of metal releases to the original content of the metal (Appendix 4.5, Section 4.5.1.3). The release of metal from sulphide minerals during oxidation is usually not a reliable indicator of the rate of acid generation due to secondary effects such as hydrolysis and precipitation of secondary minerals.

Advantages

- rates of acid generation and neutralization in a simulated moist, unsaturated, oxygenated environment can be determined;
- as an option, rates in a simulated oxygenated, fully saturated or submerged environment can be determined;
- temporal variations in rates of acid generation and neutralization can be determined;
- various control options such as blending of waste rock can be simulated;
- influence of bacteria on the rate of acid generation may be monitored;
- the test is in use in Canada and the USA and has been compared to other predictive tests to demonstrate reliability; and
- the test is relatively simple to set up and operate.

Disadvantages

- bacteria may not acclimatize to the sample;
- a long time may be required to complete test to meet program objectives; and
- interpretation of results is complex.

4.5.2.5 Columns/lysimeters

Objectives

- to determine the rate of and temporal variation in acid generation on a continuous basis;
- to determine the temporal variation in overall water quality;
- to confirm the results of static tests; and
- to test treatment/mitigation options.

Description of Test

Column tests are conducted using a variety of columnar materials, such as PVC plastic, which are filled with the sample to a certain level or with a certain weight based on the specific objectives of the program. In general, a known amount of material is placed in the column and distilled water or water-based solution is flushed through the sample or ponded on top of the sample (Figure 4.5.2-2). Water samples of the flushing water or overlying pond water is periodically collected and chemically analyzed.

Some variations are:

- wet/dry cycles;
- degree of water saturation in the sample;
- fluctuating water table;
- flushing rate through the sample;
- water forced upwards through the sample; and
- inoculation with bacteria.

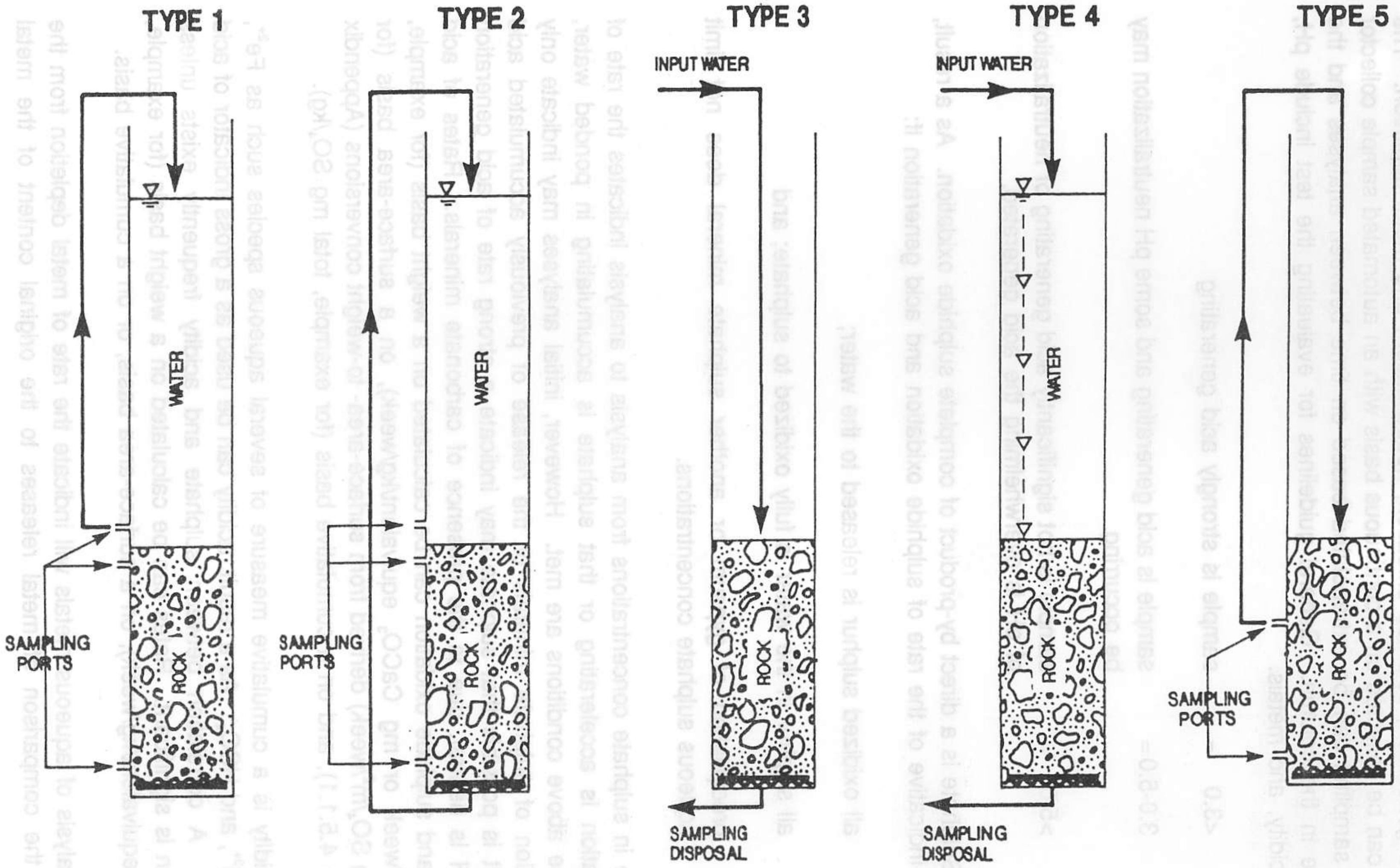

FIGURE 4.5.2-2
EXAMPLES OF COLUMN EXPERIMENTS

Interpretation of Results

The chemical analysis of water actively flushing a sample will indicate the immediate rates of acid generation, sulphide oxidation, neutralization, and metal depletion. The information can be collected on a continuous basis with an automated sample collector or periodic sampling can be extrapolated based on time between analyses and the flushing rate in the column. General guidelines for evaluating the test include pH, sulphate, acidity, and metals.

pH: <3.0 = sample is strongly acid generating

3.0-5.0 = sample is acid generating and some pH neutralization may be occurring

>5.0 = sample is not significantly acid generating or neutralization of pH is overwhelming the acid generated.

Sulphate: Sulphate is a direct by-product of complete sulphide oxidation. As a result, sulphate is indicative of the rate of sulphide oxidation and acid generation if:

- all oxidized sulphur is released to the water;
- all sulphur in the water is fully oxidized to sulphate; and
- precipitation of gypsum or another sulphate mineral does not limit aqueous sulphate concentrations.

An increase in sulphate concentrations from analysis to analysis indicates the rate of acid generation is accelerating or that sulphate is accumulating in ponded water, providing the above conditions are met. However, initial analyses may indicate only the dissolution of sulphate minerals or the release of previously accumulated acid products. It is possible that sulphate may indicate a strong rate of acid generation although pH is neutral, due to the presence of carbonate minerals. Rates of acid generation and sulphide oxidation can be calculated on a weight basis (for example, mg SO_4/kg/week or mg $CaCO_3$ equivalent/kg/week), on a surface-area basis (for example mg SO_4/m^2/week) derived from surface-area- to-weight conversions (Appendix 4.5, Section 4.5.1.1), and on a cumulative basis (for example, total mg SO_4/kg).

Acidity: Acidity is a cumulative measure of several aqueous species such as Fe^{2+}, $Fe(OH)^{2+}$, Al^{3+}, and HSO_4^-. As a result, acidity can be used as a gross indicator of acid generation. A correlation between sulphate and acidity frequently exists unless neutralization is significant. Rates can be calculated on a weight basis (for example, mg $CaCO_3$ equivalent/kg/week), on a surface-area basis, or on a cumulative basis.

Metals: Analysis of aqueous metals will indicate the rate of metal depletion from the sample by the comparison of metal releases to the original content of the metal

(Appendix 4.5, Section 4.5.1.3). The release of metal from sulphide minerals during oxidation is usually not a reliable indicator of the rate of acid generation due to secondary effects such as hydrolysis and precipitation of secondary minerals.

Advantages

- rates of acid generation and neutralization under various environmental conditions can be determined;
- temporal variations in rates of acid generation and neutralization can be determined;
- various control option such as limestone addition can be simulated;
- influence of bacteria on the rate of acid generation may be monitored; and
- variations of this test are widely used and are apparently reliable.

Disadvantages

- this test is moderately complex compared to other kinetic tests;
- test may be difficult to set up and run due to complexity of soil/water characteristics in running columns;
- bacteria may not acclimatize to the sample;
- a long time may be required to complete test to meet program objectives; and
- interpretation of results is complex.

4.5.2.6 On-site rock piles

Objectives

- to determine rate of and temporal variation in acid generation;
- to determine the temporal variation in overall water quality;
- to confirm the results of static tests;
- to test proposed control options;

- to determine the effect of on-site conditions on the rate of acid generation and water quality.

Description of Test

A relatively large quantity of the geologic unit to be tested is placed on an impervious basal drain at the minesite. The precipitation falling on the pile leaches through the sample and is directed to a container (Figure 4.5.2-3). At periodic intervals, the volume of seepage is measured and a water sample is collected for chemical analysis.

Interpretation of Results

The natural variability of climatic conditions at a site requires an interpretation that distinguishes the effects of the on-site climatic conditions from rates of acid generation, sulphide oxidation, neutralization, and metal depletion indicated by the chemical analysis of the water flushed through the rock pile. This is important because precipitation determines the flushing rate and the dilution of acid products, but may not affect reaction rates. General guidelines for evaluating the test include pH, sulphate, acidity, and metals.

pH: <3.0 = sample is strongly acid generating

3.0-5.0 = sample is acid generating and some pH neutralization may be occurring

>5.0 = sample is not significantly acid generating or neutralization of pH is overwhelming the acid generated.

Sulphate: Sulphate is a direct by-product of complete sulphide oxidation. As a result, sulphate is indicative of the rate of sulphide oxidation and acid generation if:

- all oxidized sulphur is released to the water;
- all sulphur in the water is fully oxidized to sulphate; and
- precipitation of gypsum or another sulphide mineral does not limit aqueous sulphate concentration.

An increase in sulphate concentrations from analysis to analysis indicates the rate of acid generation is accelerating, providing the above conditions are met. However, initial analyses may indicate only the dissolution of sulphate minerals or the release of previously accumulated acid products. It is possible that sulphate may indicate a strong rate of acid generation although pH is neutral, due to the presence of carbonate minerals. Rates of acid generation and sulphide oxidation can be calculated on a weight basis (for example, mg SO_4/kg/week or mg $CaCO_3$ equivalent/kg/week), on a surface-area basis (for example, mg SO_4/m^2/week) derived from surface-area- to-weight conversions (Appendix 4.5, Section 4.5.1.1), and on a cumulative basis (for example, total mg SO_4/kg).

Acidity: Acidity is a cumulative measure of several aqueous species such as Fe^{2+}, $Fe(OH)^{2+}$, Al^{3+}, and HSO_4^-. As a result, acidity can be used as a gross indicator of acid generation. A correlation between sulphate and acidity frequently exists unless neutralization is significant. Rates can be calculated on a weight basis (for example mg $CaCO_3$ equivalent/kg/week), on a surface-area basis, or a cumulative basis.

Metals: Analysis of aqueous metals will indicate the rate of metal depletion from the sample by the comparison of metal releases to the original content of the metal (Appendix 4.5, Section 4.5.1.3). The release of metal from sulphide minerals during oxidation is usually not a reliable indicator of the rate of acid generation due to secondary effects such as hydrolysis and precipitation of secondary minerals.

Advantages

- rates of acid generation and neutralization under on-site climatic conditions can be defined;
- various control options such as limestone addition can be tested;
- the test has been used in Canada and appears to be reliable; and
- influence of bacteria on the rate of acid generation may be monitored.

Disadvantages

- the test is relatively expensive and complex;
- results will vary as climatic conditions change;
- bacteria may not acclimatized to the sample;
- a long time may be required to complete test to meet program objectives; and
- interpretation of results is complex.

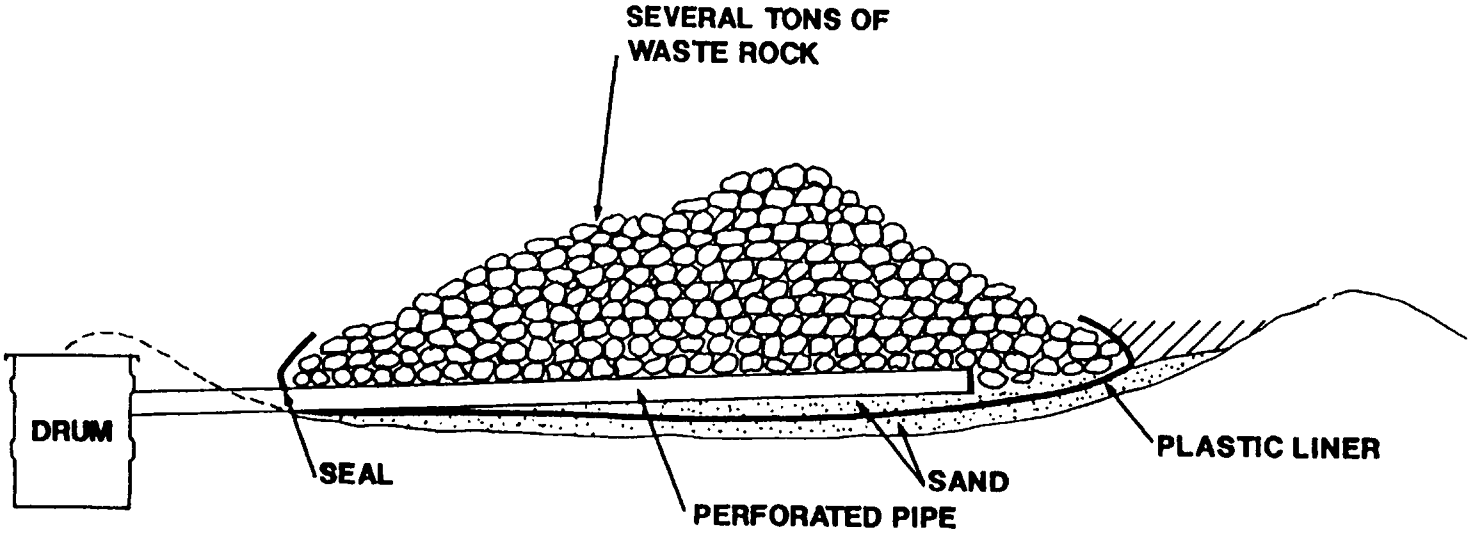

FIGURE 4.5.2-3
EXAMPLE OF A WASTE ROCK PILE