# GUIDELINE DOCUMENT FOR MONITORING ACID MINE DRAINAGE

## MEND Project 4.5.4

**This work was done on behalf of MEND and sponsored by
the Ontario Ministry of Northern Development and Mines and
the Canada Centre for Mineral and Energy Technology (CANMET)
through the CANADA/Northern OntarioDevelopment Agreement (NODA)**

**October 1997**

# GUIDELINE DOCUMENT FOR MONITORING ACID MINE DRAINAGE

Prepared For:

Mine Environment Neutral Drainage (MEND) Program

Prepared by:

Terrestrial & Aquatic Environmental Managers (TAEM) Ltd.
Saskatoon, Saskatchewan

In Association With:

SENES Consultants Limited
Richmond Hill, Ontario

PROJECT S.185
Contract 23440-5-1176/01-SQ

OCTOBER 1997

## TABLE OF CONTENTS

## ACKNOWLEDGEMENTS

The Project Manager and primary author of this document was Malcolm M$^{c}$Kee of Terrestrial & Aquatic Environmental Managers Ltd. (TAEM), with Kevin Himbeault of TAEM providing key secondary writing. Additional support was provided by Brent Topp, Gary Crosby, Alan Merkowsky, and Darryl Arsenault of TAEM with technical support for source monitoring provided by Jeff Martin and Carol Pettit of SENES. Peer review was provided by Arden Rosaasen, Peter Vanriel and Dennis Krochak of TAEM, Jeff Martin of SENES, as well as a number of anonymous reviewers selected by MEND. Graphics were completed by George Duff and editorial support was provided by Cam Irvine, both of TAEM. Members of the MEND secretariat provided constructive comments and guidance. Carl Weatherell of MEND acting in his capacity as the Scientific Authority, provided invaluable guidance and constructive criticism.

# CHAPTER I - INTRODUCTION

Acid Mine Drainage (AMD) can result when the mining process exposes mine wastes, tailings, or workings which contain quantities of reactive sulphides. Exposure of these reactive sulphides to oxygen and water may result in acid generation. The physico-chemical oxidation reactions which produce acid drainage can be further accelerated by biological activity involving bacteria. These chemical and biological reactions can generate acidic waters which increase the potential for the mobilization of contaminants (primarily heavy metals) into solution. The resultant acidic drainage waters can contain elevated metals and sulphates in addition to other acid-leached components.

Potential impacts associated with acid mine drainage are considered to be one of the greatest environmental problems faced by the mining industry today (Feasby et al. 1991). Failure to properly monitor, collect, and treat AMD can lead to contamination of groundwater and surface water resulting in the potential for significant biological degradation in the receiving environment.

The terms acid rock drainage (**ARD**) and acid mine drainage (**AMD**) are often used interchangeably. However, in this document ARD is defined as the acidic drainage resulting from both non-anthropogenic oxidation (e.g., natural outcrops and rock slide exposures) and mining related oxidation of sulphide minerals present in rock exposed to air and water. The term AMD is herein defined as acid generation from sulphide bearing rocks associated with **mining** activities.

## 1.0A OBJECTIVE

This guideline document is designed to serve as a single source introductory guide to a wide range of AMD monitoring concerns, while also providing users with information on the latest and most valuable literature sources for more site-specific concerns and emerging monitoring techniques. The document is structured to provide a guide for the design and implementation of monitoring programs from the perspective of the development of a new mine. The recognition of AMD potential and the integration of AMD monitoring within the overall environmental monitoring program during the pre-operational phase will minimize AMD effects and optimize sampling and cost efficiency. Additional information is provided for currently operating or decommissioned mines which face AMD concerns.

AMD monitoring requirements are addressed for both Source and Receiving environments, with receiving environment concerns restricted to freshwater systems. The source environment is defined as the potential freshwater contaminant pathways between the source of AMD generation and the designated point of release to the environment, which usually consists of the furthest downstream water quality compliance station. The receiving environment consists of the freshwater environment downstream (or down-gradient) of the water quality compliance stations.

## 1.0B SOMMAIRE EXECUTIF

Ce document est un guide complet qui familiarise le lecteur avec un large éventail de problèmes associés à la surveillance du drainage minier acide (DMA) et le renseigne sur la documentation la plus récente et la plus judicieuse à consulter à propos de problèmes plus spécifiques liés aux sites et des techniques de pointe. Il est conçu pour guider la conception et la mise en oeuvre de programmes de surveillance dans l'optique de l'exploitation d'une nouvelle mine. La reconnaissance des risques potentiels du DMA et l'intégration de la surveillance du DMA dans le programme global de surveillance de l'environnement avant la mise en exploitation de la mine, permettront de limiter le plus possible les effets du DMA et d'optimiser l'échantillonnage et le rapport coût-efficacité. Le document fournit également des renseignements complémentaires en ce qui a trait aux mines en cours d'exploitation ou aux mines fermées aux prises avec des problèmes de DMA.

On y traite des activités de surveillance du DMA à mettre en oeuvre tant dans l'environnement source que dans l'environnement récepteur, en se limitant, dans ce dernier cas, aux eaux douces. La notion d'environnement source s'applique aux voies d'eau douce qui risquent de véhiculer des substances contaminantes entre la source des effluents acides et le point de déversement retenu, soit généralement la station de surveillance de la qualité de l'eau située le plus en aval. L'environnement récepteur se dit des eaux douces qui se trouvent en aval des stations de surveillance de la qualité de l'eau (ou à un niveau inférieur).

## 2.0 DOCUMENT ORGANIZATION

The document is organized to fulfill a series of objectives.

1) Provide an understanding of the basic processes which generate AMD and the environmental factors which influence it.
2) Provide an overview of the theory for the development of mine monitoring programs.
3) Detail the processes for determining the potential of a mine site for acid generation.
4) Address the design considerations for developing programs for both source and receiving environment monitoring.

To meet this series of goals the document has been subdivided into four chapters and an appendix. Each section within a chapter is followed by a detailed bibliography to provide direction to the latest and most valuable literature sources for further information. A breakdown of the specific objectives of each of the chapters is provided below.

CHAPTER I, **Introduction** serves to define the objectives and describe the organization of the document.

CHAPTER II, **Basic Concepts and Principles**, consists of two sections which form the theoretical foundation for the later chapters and sections. Section 1.0 provides a general description of the geochemical/biological processes of AMD generation, and the mine site components involved. Section 2.0 establishes the basic framework for the development of monitoring programs, highlighting key concepts of monitoring theory to emphasise their importance during the completion of the practical application or execution of the monitoring components.

CHAPTER III is devoted to **Routine Source Monitoring** and is subdivided into two sections. Section 1.0 addresses design considerations for the development of a pre-operational program focussed on identifying and predicting the acid generating potential of the mining activities. This serves to identify:

- potential of the mine as an acid generating site; and
- components that will require specific AMD monitoring programs.

Section 2.0 of this chapter addresses the dynamics of AMD generation in the various sources (e.g., waste rock pile, tailings management facilities, mine workings) during the operational and decommissioning phases of a mine. The generalized AMD flow-paths from these sources to the receiving environment are detailed to provide a context for the monitoring program. Examples of generalized monitoring program designs are provided for each of the source types with additional information on the various sampling options or developing techniques.

CHAPTER IV, **Receiving Environment Monitoring** , is sub-divided into a number of abiotic and biotic sections. **Abiotic** monitoring components, consisting of Hydrology, Water Quality, and Sediments, are addressed in Sections 1.0, 2.0 and 3.0, respectively. A discussion of all the potential organisms and techniques that are available, or have been proposed as potential biological monitoring components, is beyond the scope of this document. Hence, specific **biotic** components were selected as of key importance to **most** mining operations for the reasons outlined in the introduction to CHAPTER IV. The selected primary (Tier I) biological components involve the monitoring of benthic macroinvertebrate communities and a range of fish indices (population, health, and bioaccumulation). Additional biological monitoring options are summarized and/or appropriate literature sources are provided in the section bibliographies.

These sections have been deliberately structured to emphasize the importance of developing the basic foundation and principles of the operational monitoring program PRIOR to executing intensive baseline Investigations. This is the only way to ensure that baseline data adequately supports the operational and decommissioning monitoring program. Hence, the sections follow a sequence consistent with the developmental phases of a mine. Each of the sections commences with an introduction of the specific monitoring component and its relationship to AMD. This is followed by the key factors to be addressed or considered when using that monitoring component during the monitoring stages. These components are listed below in sequential order.

- Environmental characterization requirements.
- Development of the monitoring program.
- Completion of the intensive baseline.
- Implementation of the refined operational monitoring program.
- Completion of the decommissioning baseline.
- Implementation of the decommissioning monitoring program.

The final section of CHAPTER IV provides a basic introduction to the role of toxicity testing in a mines overall environmental monitoring plan.

A series of **Technical Summary Notes** (TSN) is provided in an accompanying appendix. These technical summary notes detail information on specific sampling techniques which are cross-referenced in the text of the main document. These notes are designed to provide basic summary information on specific sampling tools or protocols, addressing the advantages and disadvantages of the technique(s), and, whenever possible, an assessment of the cost efficiency is provided. A list of recommended source documents is included for each summary note.

# CHAPTER II - BASIC CONCEPTS AND PRINCIPLES

## 1.0 DESCRIPTION OF ACID MINE DRAINAGE PROCESSES

### 1.1 Chemical Processes for Acid Generation

Sulphide minerals are crystalline substances which contain sulphur in combination with a metal (e.g., $FeS_2$) or a semi-metal (e.g., AsS) or both (e.g., FeAsS) without oxygen (Table II:1.1-1). Acid mine drainage (AMD) refers to the contaminated discharge which results when sulphide-bearing mine-wastes undergo a natural oxidation process leading to the production of acid and dissolution of oxidation products including metals. This oxidation process involves both the chemical and biological oxidation of sulphide minerals such as, but not limited to, pyrite ($FeS_2$), marcasite ($FeS_2$), pyrrhotite ($Fe_{1-X}S$), and chalcopyrite ($CuFeS_2$). As an example, consider the production of acid resulting from the chemical oxidation of **pyrite** as indicated in the following stoichiometric equations.

$$FeS_2 + \frac{7}{2} O_2 + H_2O \rightarrow Fe^{2+} + 2\, SO_4^{2-} + 2\, H^+$$

$$2\, FeS_2 + 7\, O_2 + 2\, H_2O \rightarrow 2\, FeSO_4 + 2\, H_2SO_4$$

or

$$2\, Fe^{2+} + \frac{1}{2} O_2 + 2\, H^+ \rightarrow 2\, Fe^{3+} + H_2O$$

Aqueous ferrous iron ($Fe^{2+}$) is oxidized to its ferric state as follows:

The reaction given by Equation 3.3 is dependent upon the pH of the solution and presence of microbial catalysts such as *Thiobacillus ferro-oxidans* and other acidophilic bacteria (Table II:1.1-2). Under acidic conditions (pH 2 to 3), the biological oxidation rate is approximately 16 to 35 times greater than the chemical rate.

The sensitivity of the dissolved ferric iron to variations in pH is shown in Figure II:1.1-1. Little ferric iron remains in solution at pH 2 to 3; instead it is hydrolyzed to ferrihydrite $Fe(OH)_3$.

$$Fe^{3+} + 3H_2O \rightarrow Fe(OH)_3 + 3H^+$$

Under low pH conditions, available ferric iron acts as an oxidizing agent to produce additional

$$14\,Fe^{3+} + FeS_2 + 8\,H_2O \rightarrow 15\,Fe^{2+} + 2\,SO_4^{2-} + 16\,H^+$$

sulphate and hence sulphuric acid. The anoxic oxidation of pyrite is as follows:

$$FeS_2 + \frac{15}{4}O_2 + \frac{7}{2}\,H_2O \rightarrow Fe(OH)_3 + 2\,H_2SO_4$$

The overall equation for the previous reaction is:
As illustrated by the above reaction, one mole of pyrite can produce two moles of sulphuric acid. This acid may combine with water percolating through the mine waste and lead to the production of contaminated discharge through dissolution of metals and reaction.

Sulphide minerals other than pyrite may have different reaction mechanisms, stoichiometries, and reaction rate limiting factors. For instance, the oxidation reaction for pyrrhotite is as follows:

$$Fe_{1-x}S + \frac{9}{4}\,O_2 + \frac{5}{2}\,H_2O \rightarrow Fe(OH)_3 + H_2SO_4$$

As indicated by equation 3.7, one mole of pyrrhotite can produce one mole of sulphuric acid. The chemical composition of sulphide minerals varies due to such factors as ionic substitutions, alteration, and pressure-temperature conditions at formation. Predictive capabilities requires an understanding of the types and distribution of sulphides within mine wastes, and a knowledge of the chemical and physical properties of the sulphides which affect the rate of oxidation reactions and, hence, acid production.

Acid produced by sulphide oxidation may be naturally neutralized by minerals such as carbonates, oxyhydroxides, or aluminum-silicates (Table II:1.1-3). Examples from these three classes of minerals are calcite ($CaCO_3$), gibbsite ($Al(OH)_3$), and albite plagioclase feldspar ($NaAlSi_3O_8$). The most common acid consuming mineral is calcite ($CaCO_3$) which consumes acidity through the production of $HCO_3^-$ or $H_2CO_3^0$ as shown below:

$$CaCO_3 + H^+ \rightarrow Ca^{2+} + HCO_3^-$$

and

$$CaCO3 + 2\,H^+ \rightarrow Ca^{2+} + H_2CO_3^0$$

The ratio of sulphide minerals to acid consuming minerals is the determining factor concerning whether the rock will eventually become acid generating.

## 1.2 Stages of AMD Production

AMD arises when conditions permit rapid oxidation, which generates sufficient acid to eventually consume the neutralizing potential of the surrounding material leading to the release of contaminated waters from the source. There are three stages in the overall lifespan of an acid mine drainage as indicated in Figure II:1.2-1.

In Stage 1, the chemical oxidation of sulphides begins as the exposed surfaces of sulphide minerals are oxidized through reaction with atmospheric oxygen (Eq. 3.1 & 3.2). The ionic products of this initial reaction include acidity ($H^+$), sulphate ($SO_4^{-2}$), and ferrous iron ($Fe^{2+}$). Ferrous iron ions may be further oxidized to ferric iron ($Fe^{3+}$) (Eq. 3.3). The loss of ferric iron from solution (Eq. 3.4) assists in limiting the rate of the initial oxidation process. When present, calcium-based buffering materials such as calcite can buffer the acid produced. Drainage from sulphide containing wastes where oxidation reactions are in Stage I typically has neutral or alkaline pH, and low to elevated sulphate concentrations. Sulphate concentrations are limited by the solubility of gypsum ($CaSO_4$ $2H_2O$) at neutral range pH.

In Stage 2, the reaction rate increases in response to depletion of the buffering capacity and/or microbial catalysis. Drainage from reactive mine wastes at this stage is characterized by:

- a progressive decline in pH to levels determined by the buffering mineralization and capacity available;
- increased concentrations of ferrous iron and sulphate;
- decreasing solution alkalinity; and
- increased metal concentrations.

In Stage 3, any remaining available buffering capacity is consumed and the microbial catalyzed oxidation of the remaining sulphide minerals controls the oxidation rate. Ferrous iron can be oxidized to ferric iron at a high rate thereby accelerating the acid production process. The rate of microbial catalysis is a function of temperature, solution pH, nitrate, ammonia, phosphorous concentrations, and the population density of the bacteria. Drainage from reactive mine wastes during Stage 3 is characterized by low pH, and elevated concentrations of sulphate and metals.

At the end of Stage 3, exposed sulphides will be exhausted or otherwise become unavailable for oxidation, and acid production will cease. The acid production life of reactive mine wastes can vary widely from less than one month to several centuries, depending on physical, chemical, and microbiological conditions. Additional details on these acid generating reactions are available in the literature listed in the bibliography.

AMD is typically composed of a mixture of water that has been contaminated with oxidation reaction products and uncontaminated water (e.g., run-off). Hence, AMD quality varies in response to infiltration of “clean” water (e.g. precipitation) which influences oxidation, dilution, and flushing rates.

### 1.3 AMD Sources

A ranking of the AMD potential for seven of the more common types of ore deposits in western Canada is provided in Table II:1.3-1. The relative tendency for these mineral deposits for AMD is predominantly determined by the net-acid generating potential of the prevalent mineral assemblages, accessibility of the reacting minerals to weathering agents, and potential mediation by iron-oxidizing bacteria (MEND 1.32.1 1993).

Sulphide oxidation reactions commence when there are sufficient quantities of oxygen and water available. Mining activities create these conditions wherever sulphides are exposed to the atmosphere.

The three main potential sources of acidic drainage at mine sites are:

1) broken waste rock or ore;
2) tailings; and
3) mine workings and other exposed rock surfaces (e.g., pit walls and pit floors).

The first category is the most diverse area of concern as rock removed during mining may be present at a minesite in a number of forms such as waste rock dumps; ore stockpiles; low grade or marginal ore stockpiles; heap leach piles; and as waste fill used in the construction of parking lots, yards, building foundations, and roads.

Tailings facilities can be a significant source for long-term acid generation. Hence, facilities should be designed for long-term management/prevention of AMD during the decommissioning phase. The management of acidic drainage associated with mine workings should be factored into the mine water management plan during the operational phase with long-term AMD management/prevention factored into the decommissioning phase.

### 1.4 The Role of Climatic Factors

The peak loads of AMD discharge tend to exhibit seasonal patterns with the exception of continuously pumped adit waters (BC AMD 1990c). During dry periods (low precipitation or freezing conditions), enough water may be present to support the AMD oxidation process but the infiltration rate may not be sufficiently high to flush these acid waters and associated contaminants produced from the source. This results in the *in situ* accumulation of acid and metal salts with little evidence of AMD contamination in the released surface waters. The flushing of these accumulated contaminants by heavy rains or snow melts, can release a concentrated slug of high acid/metal contaminated water. This seasonal pattern is evident in the annual hydrograph and corresponding copper and zinc concentrations shown in Figures II:1.4-1 and II:1.4-2 for a coastal mine and an inland mine, respectively.

Subsequent rains or melts may result in the flushing of equal or higher concentrations if salts remain from previous infiltrations, or of lower concentrations if the earlier washings were thorough. This variable flushing effect means that, despite the overall seasonal pattern in the release of AMD contaminants, there is often a poor correlation between AMD contaminant concentrations and corresponding discharge measurements. This is due to the sensitivity of the flow/concentration relationship to antecedent flow conditions (i.e., whether discharge has been increasing, stable, or decreasing). As previously described, following freezing or dry periods, the first waters to be released from the toe of a rock pile may contain high contaminant concentrations. These tend to increase with discharge as rock surfaces are thoroughly flushed. If the infiltration rate stabilizes contaminant concentrations may remain stable or may begin to decline as the available soluble salt reserves are depleted. Eventually, concentrations will decline independent of the rate of discharge, whether increasing, stable or decreasing, as the salt reserves in the source are depleted. At the end of a single hysteresis loop the water flushing from a source will only contain AMD generated during that flushing period. This **hysteresis effect** is illustrated in Figure II:1.4-3 and has important implications for water quality monitoring which are discussed in ChIII:2.0, and ChIV:1.0 and 2.0.

### 1.5 Section Summary

Acid mine drainage (AMD) refers to the contaminated discharge which results when sulphide bearing mine wastes undergo a natural oxidation process leading to the production of acid and dissolution of oxidation products including metals.

Three stages occur in the overall lifespan of AMD:

| | |
|---|---|
| Stage 1 | chemical oxidation of sulphides begins as exposed surfaces of sulphide minerals react (oxidize) with atmospheric oxygen; |
| Stage 2 | the reaction rate increases in response to depletion of the buffering capacity and/or microbial catalysis; and |
| Stage 3 | any remaining available buffering capacity is consumed, and microbial catalyzed oxidation of the remaining sulphide minerals controls the oxidation rate. |

At the end of Stage 3, exposed sulphides will be exhausted or otherwise become unavailable for oxidation, and acid production will cease.

The three main sources of acidic drainage at minesites are:

1) broken waste rock or ore;
2) tailings; and
3) mine workings and other exposed rock surfaces (e.g., pit walls and pit floors).

During periods of low precipitation or freezing, *in situ* accumulation of acid and metal salts may occur with little evidence of AMD contamination in the released surface waters. When these accumulated contaminants are flushed by heavy rains or snow melts, a concentrated slug of high acid/metal contaminated water can be released. There is often a poor correlation between AMD contaminant concentrations and corresponding discharge measurements. Concentrations will decline independent of the rate of discharge, whether increasing, stable or decreasing, as the salt reserves in the source are depleted. This is known as the hysteresis effect which has important implications for water quality monitoring.

## 1.6 Section Bibliography

**The Process of Acid Generation**
BC AMD 1989
BC AMD 1990a
Helz et. al. 1987
Kwong 1991
Lowson et al. 1991
MEND 1.32.1. 1993
Morin and Cherry 1988
Morin et al. 1988

**Bacteria and Acid Generation**
Lundgren and Silver 1980
Wallis and Ladd 1984.
Fortin et al. 1995
Schippers et al. 1995

**Climatic Factors and AMD**
BC AMD 1990c
MEND 3.22.1 1994
Nordstrom et al. 1990

## 2.0 DEVELOPMENT OF MONITORING PROGRAMS

The goal of this section is to identify the basic framework and design of monitoring programs and to emphasize the role that specific AMD monitoring plays in the overall environmental monitoring plan for a mine. This chapter is not meant to be a treatise on monitoring theory, but is provided to identify the role that each of the separate monitoring components (whether source or receiving environment) play in the overall monitoring program. Prior to developing a specific monitoring program, readers should familiarize themselves with the recommended literature listed in the section bibliography.

## 2.1 Introduction

Effective monitoring programs are designed to detect potential environmental impacts before irreparable damage results in the receiving environment. Monitoring programs should be developed within a clear framework, with well defined objectives. The most efficient environmental monitoring plan is one that incorporates all of a mine's environmental concerns, such that each component supports and builds on the others. This will prevent the monitoring program from deteriorating into a self-perpetuating data collection exercise which fails to meet the original site-specific program objectives.

There are three basic stages of development for a mining operation:

1) pre-operational phase;
2) operational phase; and
3) decommissioning phase.

The basic framework of any monitoring program should parallel these phases as shown in Figure II:2.1-1. Each of these phases is discussed in more detail in the relevant sections to follow.

## 2.2 Pre-Operational Phase

The pre-operational phase for a mine is an important period for the development and long-term relevance of a monitoring program. Failures in design and execution during this stage of the monitoring program will resonate throughout the monitoring lifespan of the mine with significant negative implications for cost efficiency and environmental safety.

During the pre-operational mining phase there are three main objectives for the monitoring program (Figure II:2.1-1):

- **finalize the mine plan** to minimize environmental impacts;
- **design the monitoring program** to detect potential impacts; and
- **collect the intensive baseline data** to support the monitoring program.

### 2.2.1 Development of Final Mine Plan

Development of a Final Mine Plan incorporates two lines of investigation:

- Project Characterization/Preliminary Impact Identification; and
- Preliminary Environmental Characterization.

These two processes tend to be completed concurrently with the results combined to develop the final mine plan.

### 2.2.1.1 Project Characterization/Preliminary Impact Assessment

Project characterization identifies the physical and structural components of the project (e.g. geology, infrastructure, etc.) which have the potential to impact the environment. The portions of the study area to experience physical disturbance (infrastructure and mine placement) or effluent inputs are identified. This procedure highlights potential impact areas and the boundaries for interaction between the source and receiving environments.

Project characterization identifies:

- the ore type(s) and associated host lithologies;
- the proposed mining method;
- the proposed ore processing or milling procedures;
- proposed sites for surface structures; and
- the potential or proposed points of release to the environment.

The objective is to identify and catalog potential **sources** of disturbance or contaminant release to the environment. Once the project characterization is complete, the next step is to make a preliminary assessment of the impacts and devise means to minimize such impacts. The following three subsections should be considered.

#### *Prediction of AMD Potential*

Identifying the potential AMD impacts to the environment involves determining the potential of the minesite for acid generation. This predictive program should serve to:

- identify the different geological units that have potential for AMD;

- meet information requirements for decisions on the destination and placement of waste rock;
- estimate the time to AMD generation; and
- identify potential receiving environments.

A schematic showing the role of predictive AMD assessment in the development of the final mine plan is shown in Figure II:2.2-1. Predictive AMD assessment should be completed for each of the represented lithological units of the mine development plan, as each unit may have a different AMD potential. Should the predictive tests provide **NO** evidence of AMD potential the next step in the flow diagram is bypassed. Details on the procedures to follow for predicting AMD potential are provided in ChIII:1.0.

### *Re-Assess Waste Management Plan*

Should the predictive program identify potential AMD sources, then:

- waste management plans should be re-assessed accounting for AMD; and
- flow paths should be developed for each potential AMD source, by
    - identifying pathways from the **source** to the **receiving** environment.
    - completeing impact/risk assessment modelling for each identified pathway.

### *Identify Non-AMD Impacts*

The non-AMD potential impacts (e.g., mill and dewatering effluent) are identified along with their probable pathways based on the project characterization as was done for the potential AMD impacts.

#### 2.2.1.2 Preliminary Environmental Characterization

Preliminary environmental characterization, sometimes referred to as preliminary or reconnaissance environmental surveys, involves the characterization of the environment associated with the mining project. This consists of:

- characterization of the abiotic components
    - e.g., climate, hydrology, water quality, sediments, and
- characterization of the biotic components
    - e.g., benthic macroinvertebrates, plankton, fish.

The preliminary survey identifies rare species, valuable and potentially sensitive organisms or habitats, and potential sampling stations (reference/control and impact) for design of the monitoring program.

#### 2.2.1.3 Final Mine Plan

The results of the *Preliminary Environmental Characterization* and the *Project Characterization/Preliminary Impact Identification* investigations are incorporated into the development of the final mine plan. The potential impact pathways and impact/risk assessments are refined by incorporating specific indicator organisms and/or communities identified in the receiving environment. Modelling of each identified pathway can be completed using the site specific information generated by the preliminary environmental characterization.

Based on the modelling results, the proposed operating and waste management procedures can be optimized to minimize associated environmental impacts in the most cost-efficient manner. Once the operational and waste management procedures have been optimised to control AMD and non-AMD impacts, the final modelling and impact/risk assessment is completed. Such a pro-active approach will minimize the risk of accruing expensive environment-related costs in the future.

### 2.2.2 Operational Monitoring Program Development (Figure II:2.2-2)

In the following discussions, the term **components** is used in reference to the physical and biological constituents of the environment that form sampling compartments of the monitoring program. Examples of the most commonly incorporated components of monitoring programs are:

| <u>Physical Components</u> | <u>Biological Components</u> |
|---|---|

- Water Quality (surface and groundwater)
- Hydrology (surface and groundwater)
- Sediments

- Plankton
- Benthic macroinvertebrates
- Macrophytes
- Fish

Interactive Components

- Habitat Classification
- Toxicity Assays

The term **parameters** is used in reference to measurements collected from these components. In this context parameters would include selected chemical (e.g., trace metals) / biochemical (e.g., detoxifying enzymes) constituents measured in water, sediment, or fish tissue samples, population or community measurements determined for the biological components, and selected end points for toxicity assays.

#### 2.2.2.1 Identifying and Classifying Monitoring Requirements

The completion of the final mine plan provides the necessary information to develop and refine the operational monitoring program. Each monitoring requirement of a project should be classified as one of the following monitoring types:

***Compliance Monitoring***

- Is dictated by the regulatory authorities through permits as a condition for permission to develop, operate, or decommission a site.
- Generally focuses on points of discharge into the receiving environment.
- Has specific protocols such as sampling frequency, scope of analysis, and stations detailed in the operating permit.

***Diagnostic Monitoring***

- Is routine monitoring generally completed on the initiative of the operator.
- Focuses more on the potential sources of AMD than on the downstream compliance monitoring stations.

- Serves as an early warning system of potential shifts towards increased AMD generation, thereby providing an operator response time prior to an actual non-compliance event.
- Study design is determined by the operator and based on the operator's needs.

An example of a diagnostic monitoring component would be the monitoring of the drainage from a waste rock pile to provide early warning of acid generation upstream of a compliance station.

#### *Investigative Monitoring*

- Initiated by the operator or required by regulatory agencies in response to non-compliance results, upset events (e.g. spills), or operational design changes.
- **Not** considered part of the compliance or routine diagnostic monitoring programs.
- Study design determined by the operator or developed through discussions between the operator and the regulatory agencies on the requirements of the program.

Investigative monitoring can range from simple surveying of a site to assess the integrity of an old culvert for a waste rock drainage ditch, to a complex investigation of the bioavailability of trace metals in the sediments of a subaqueous tailings disposal site. Investigative monitoring addresses site-specific operational and environmental concerns which are beyond the scope of this document.

### 2.2.2.2 Tiering Concepts

It is neither logistically feasible, nor cost effective, to sample all possible physical and biological components and their respective parameters. An efficient monitoring design incorporates monitoring **tiers** based on trade-offs between a component/parameters value as an early warning indicator, and the expense associated with the use of the component/parameter. Tiering is based on the concept of initially using a set of simple tests, with the tests increasing in complexity and costs only if early results identify a need for more information (Hodson et al. 1996). When evidence, or effects of AMD are identified by simple Tier I tests, more complex tests (Tier II) should be initiated to confirm the evidence and determine the general source and spatial distribution. Tier III tests are then initiated to determine the causal mechanisms of the effect in order to identify a long-term solution to the problem. The following subsections outline the tiering of monitoring components, parameters, and sample frequency. The importance of tiering decision rules is also discussed. Proposed tiering structures designed to meet the monitoring requirements for a generic mine site are outlined as a preface for each of Chapters III and IV (Source and Receiving Monitoring, respectively).

### *Tiering of Monitoring Components*

The number of potential components available for inclusion in a monitoring program is extensive, ranging from recognised standard components, such as surface water and bulk sediment chemical analyses, to more complex and costly components, such as *in situ* pore water chemical analysis and sediment toxicity assays. Classification of components into specific tiers is site and component specific.

### *Tiering of Parameters*

Another form of tiering involves increasing the number of parameters measured within a component as environmental concerns increase. For example, diagnostic monitoring of the water in a waste rock pad drainage ditch could involve the tiering of the measured parameters as shown below.

- Tier I: pH and sulphate concentrations
- Tier II: trace metals measured if a decrease in pH and/or an increase in sulphate concentrations (possible AMD production) become evident.

### *Tiering of Sample Frequency*

A monitoring program may also be designed to incorporate tiering into the sampling frequency. For example, exceedance driven, or Markovian sampling designs are often used for water quality monitoring. These monitoring designs incorporate increase sampling frequency in response to previous analytical results from the monitoring program.

### *Tiering Decision Rules*

A key component of any design involving tiering is defining the point at which to initiate the additional tiers. Data from the Tier I components/parameters must be interpreted conservatively if they are to serve as early warning indicators.

### 2.2.2.3 Theoretical Study Designs

Monitoring study designs may be separated into two categories:

1) sentry designs; and
2) impact designs.

### *Sentry Designs*

Sentry monitoring designs serve as early warning systems to prevent violation of licence requirements. Compliance monitoring requirements and most diagnostic monitoring programs use sentry monitoring designs. They generally have the following characteristics:

- compliance or diagnostic monitoring programs that involve repeated sampling over time;
- parameter of interest acts as a proxy, signalling when further, more intensive monitoring or investigative programs should be initiated;
- reference (control) stations are preferable but not mandatory; and
- do not provide direct conclusions on environmental impacts.

Most source monitoring programs consist of sentry monitoring designs.

### *Impact Designs*

The objective of impact monitoring is to determine whether a measurable impact has occurred in the receiving environment as a result of mining activities. Impact monitoring programs involve one of four basic designs:

1) Control-Impact (**CI**) or Spatial Design;
   - spatial comparisons between potentially impacted sites and reference sites.
2) Before-After (**BA**) or Temporal Design; and
   - temporal comparisons usually between pre-impact (baseline) and impact data.
3) Before-After-Control-Impact (**BACI**) or Spatio-Temporal Designs.
   - these provide both spatial and temporal comparisons;
   - they can ability to investigate the interactions of time and space;
   - BACI designs are the more rigorous of the three designs.
4) Gradient Design
   - spatial comparisons along an identifiable contaminant gradient.

Study designs incorporating control stations (C-I and BACI) are further enhanced if more than one reference station is used. Multi-control designs (Underwood 1994) employ several reference stations to differentiate more reliably between real impacts and temporal/spatial noise. The increased costs of multiple reference stations are compensated by the reduced risk to the operator of a measured change in a monitoring component being incorrectly designated as evidence of mining impacts. This would result in the initiation of additional unnecessary and costly diagnostic or investigative monitoring programs.

Investigations of upset or unanticipated events, such as spills, are often restricted to CI designs due to a lack of baseline data. In the absence of control stations[1] and/or baseline data, gradient study designs may be used. These involve investigations along a downstream gradient increasing in distance from the point of release. In such studies impacts are inferred from the spatial distribution of the measured parameters (e.g., high near source, progressively lower downstream and correlated with observable effects).

### 2.2.2.4 Monitoring Model Parameters

Modelling plays an important role in AMD risk and impact assessment. Mine monitoring programs should provide information feedback to determine the accuracy of the models and to test the suitability of the assumptions and selected input parameters. Monitoring feedback for model assessment will be site and model specific. Important modelling considerations for monitoring program design include:

- sensitivity analysis to identify the key input parameters which should be incorporated into the monitoring program; and
- the integration of model output parameters (or predicted results) into the monitoring program to compare observed impacts to predicted impacts.

Incorporation of monitoring feedback into risk and impact modelling improves the predictive power of the models and therefore their value in the future evaluation of decommissioning alternatives. Monitoring components and parameters will be model and site-specific.

### 2.2.2.5 Sampling Considerations

---

[1] Can occur when the effluent release point is actually the headwater of a stream.

Once the study design for each monitoring component has been determined, the specific sampling details for the intensive baseline program are identified using the information collected in the preliminary environmental characterization program. These involve:

- determining sampling station locations with a minimum of one station upstream of the point of compliance as an advance warning station (use a multi-control BACI design with near and far field[2] stations whenever possible);
- identifying specific parameters; and
- identifying appropriate sampling equipment and protocols.

These decisions are primarily specific to each of the selected monitoring components. Basic statistical considerations are addressed in ChII:2.5. More detailed information on sampling procedures/requirements for the most common monitoring components are addressed in the component specific sections in Chapters III and IV.

Once the study design of the operational monitoring program has been determined, intensive baseline sampling can be conducted in order to provide the temporal control to support the future operational monitoring program.

### 2.2.3 Complete Intensive Baseline Investigation

It is important that the **Intensive Baseline** investigation be initiated only after the study design has been finalized. Collection of baseline data prior to the planning of the monitoring program often leads to the inclusion of inappropriate data with associated unnecessary costs. At a minimum, the intensive baseline survey should be treated as the first cycle of the monitoring program and involve the following steps:

- sampling monitoring stations for each of the Tier I **components**;
- sampling monitoring stations for each of the Tier II **components** that require baseline data support;
- sampling protocols identical to those proposed for the monitoring program; and

---

[2] In aquatic systems, near field refers to an area in an impacted drainage system which is immediately downstream and far field refers to an area removed by a distance from the point discharge (further downstream). Specific determination is often based on percent dilution of the contaminant plume.

- measurement of all proposed **parameters** (Tiers I, II, and III).

In some cases these investigations may have to be far more intensive than the proposed operational monitoring program, in order to identify any in-system dynamics which need to be incorporated into the monitoring design (see ChII:2.5). An example of this is the intensive preliminary investigations required to adequately design a water quality program for an acid generating source (see ChIII:2.5). Ideally, both the initial environmental characterization and the intensive baseline program should incorporate a minimum of two full seasons of pre-operational data.

## 2.3 Operational Phase

During the operational phase the monitoring program is initiated as shown in the flow chart in Figure II:2.3-1.

- Complete the sampling of Tier I components each cycle.
- Evaluate Effects
  - No Effect: monitoring schedule shifts to the next monitoring cycle.
  - Possible Effect: Initiate Tiering
    Analyse Tier II parameters for the core component;
    e.g., analyse metals in addition to pH and sulphates in water samples.
    Initiate secondary sampling components;
    e.g., sediment toxicity assays.

- Assess results of Tier II parameters or components.
  - No evident impact: Shift to next cycle but operate at second tier.
  - Evidence of impact: Complete Risk Assessment
- Complete a Risk Assessment: Are follow-up actions required?
  - No: Go to next monitoring cycle.
  - Yes: Implement follow-up actions (Tier III and/or mitigative actions).
    Revise monitoring program to support follow-up actions.

It is important that periodic reviews of the program (e.g., Status of the Environment reviews) be completed to ensure that the monitoring program is meeting the operational and regulatory needs of the project. This periodic review also serves to assure that the data are being collected and stored in a manner which can be readily assessed and analysed. These reviews act as impact assessments over the operational period to date, which may result in the detection of trends not readily apparent in year end reviews and serve as a test of the accuracy of the predicted impacts as proposed in the Environmental Impact Assessment.

## 2.4 Decommissioning Phase

The decommissioning phase of any monitoring program focuses on two different types of projects:

1) recovery; and
2) control.

### *Recovery Monitoring*

- Confirming that impacted (e.g. on-site or near site) habitats are exhibiting recovery with the cessation of the mining activities.
- Determining whether additional remedial action is required or whether environmental processes will result in system recovery with time.

### *Control Monitoring*

- Monitoring the integrity and operation of any control structures or procedures (e.g., subaqueous tailings disposal, dry covers) to ensure that potential AMD production is restricted or isolated within the control structure.

Decommissioning monitoring tends to decrease in intensity and frequency over time as:

- evidence of recovery in the impacted systems accumulates; and
- confidence in the security of the control structure is achieved.

Monitoring will cease when sufficient data are available to support the conclusion that the decommissioning criteria, agreed on between the operator and the regulatory agencies, have been met.

### 2.4.1 Pre-Decommissioning Baseline

Pre-decommissioning baseline serves the same purpose as the pre-operational baseline. The objective is to classify and characterize the state of the potential AMD sources and their respective receiving environments immediately prior to, or after, closure. Impacted sites should be identified and characterized in order to monitor for recovery during the decommissioning phase.

The pre-decommissioning baseline should be designed to identify any remedial action required to promote the recovery of the on- and near-field impacted habitats. In control monitoring the pre-decommissioning baseline should adequately identify the most appropriate monitoring program (e.g., station locations, sampling frequency) to determine whether the control structure is functioning as designed.

## 2.5 Statistical Considerations

This section addresses some of the basic statistical concepts that should be considered when designing a monitoring program. The discussions are restricted to those concepts that arise in the sections to follow. The objective is to provide the **minimum** required background to emphasise certain key considerations in developing a study design and is not a suitable replacement for the recommended papers and texts listed in the bibliography following this section. The importance of a statistical approach to study design cannot be over-emphasized. It is recommended that a biostatistician review monitoring designs during their development phase.

### 2.5.1 Sample Variation

There are essentially three forms of sample variation: spatial, temporal and instantaneous. **Spatial** variation (among stations) refers to the measurable differences between sampling locations. Temporal and instantaneous variations occur **within** a station, with **temporal** referring to variation between sampling periods and **instantaneous**[3] relating to variation in samples collected during the same sampling period.

---

[3] Instantaneous is a relative term. e.g. 5 benthic samples collected at a station within a few hours may be considered instantaneous sampling as significant changes in the community would not be expected within the replicate sampling intervals (see autocorrelation).

The use of single samples to represent a population at a specific sample station for a single sampling period assumes that there is no **instantaneous** or **spatial** variation. For example, often a single monthly water sample is collected to represent the average concentration for that month. How representative that sample is of the actual monthly mean is dependent on how variable the concentration was over the month. The more the concentrations vary over the month, the less representative of the actual mean is the single value.

The collection of single samples without an understanding of the variation inherent in the system is of little management value other than for long-term trend analysis. If rapid response management decisions are required, information on the reliability of specific measurements is essential. To illustrate, consider a mine site with two waste rock drainages (streams A and B), each with a permitted maximum zinc concentration of 1.0 mg/L. At the diagnostic station for stream A, the single water sample concentration was 0.9 mg/L while in stream B the sample was 0.7 mg/L. Without information on the inherent variability within each of these diagnostic stations, a manager cannot anticipate which stream is at risk of exceeding permitted zinc concentrations at its compliance station. However, if the variance was known to be low at Station A (e.g., SD=± 0.01g/L), and high (e.g. SD=± 0.4) at Station B, it becomes clear that the station of concern is Station B, despite a lower Zn concentration.

Differences between sites (spatial variation) can only be determined when differences within a site (instantaneous variation) are understood. For example, five replicates were collected at two sediment stations progressively downstream from a mine mill effluent release point (TAEM 1996). The mean nickel concentration upstream was 43.4 • g/g while the downstream station mean was 25.6 • g/g. Despite the apparent difference between these two concentrations, the 95% confidence intervals for these stations (±16.8 and ±11.024 • g/g, respectively) overlap and, hence, we cannot be confident that the apparent difference between the two sites is real. Had only single samples been collected two conflicting worst case interpretations may have resulted:

1) apparently greatly different nickel concentrations (58 and 10 • g/g, respectively), or
2) apparently similar concentrations (27 and 25 • g/g, respectively).

### 2.5.2 Sample Replication

Replicated sampling is required in preliminary investigations to identify the inherent variation within a system. Replicated sampling must also be incorporated into the actual monitoring program when accurate means rather than peak values are the required objective. Once a measure of the variance within a system has been determined by a preliminary investigation the number of replicates required to achieve the desired precision for an estimated mean can be determined. This is most commonly achieved through **Power Analysis**, which is discussed in greater detail in the references listed in the section bibliography.

### 2.5.3 Frequency Distribution

The nature of the frequency distribution of a data set has significant implications for study design and data analysis. The frequency distribution may serve to identify irregularities or patterns in the data set with implications for sample design. It also provides information on the nature of the distribution (e.g. normal, bivariate, etc.) and, hence, the most appropriate statistical procedures (transformations, parametric or non-parametric, Monte Carlo techniques, etc.) for analysis.

Consider the changes in concentration of a specific water parameter sampled repeatedly (intensive sampling program) over a specific time period. The possible time plots and their corresponding frequency distributions are provided in Figure II:2.5-1 with examples discussed below.

The concentration of a specific parameter in groundwater or the effluent of a rigidly controlled treatment process may remain relatively constant over time similar to Figure II:2.5-1a. Such systems may be accurately monitored using single samples if the variance can be expected to remain low and stable. Hourly surface temperature measurements may conform to a normal distribution as shown in Figure II:2.5-1b, where the data are symmetrically distributed around the mean (BC AMD 1990c). Parameters with normal distributions are ideally suited to analysis using parametric statistical procedures. Skewed distributions (Figure II:2.5-1c), often censored by a detection limit (Figure II:2.5-1d), are common to water quality data and indicate that data transformation or non-parametric analytical procedures must be used. Bi-modal or multi-modal distributions (Figure II:2.5-1e) are also common to water quality data when results from time periods with different distributions (e.g. seasonal influences) have been grouped together (BC AMD 1990c).

### 2.5.4 Stratification

When a sampling program fails to incorporate temporal (e.g., seasonal discharge patterns) or spatial (e.g. substrate composition) factors which influence the mean and variance of a parameter, the variance in the data set increases. These influences are called strata and are often visibly evident as bimodal or multi-modal frequency distributions (Figure II:2.5-1e) and are of significant concern when monitoring AMD water quality. Leaving the data grouped would greatly increase the uncertainty associated with each sample, and therefore greatly increase monitoring sample requirements. This may be prevented by introducing some form of stratification in the sampling program.

In order to optimize a sampling program a preliminary investigation[4] is required to identify any strata where the measured parameters have substantially different means and variances. Redistribution of the sampling effort so that all strata are sampled with equal efficiency will result in equal confidence intervals for the strata. The result is an economical sampling design where sampling effort is concentrated when (or where) it will provide the greatest value (BC AMD 1990c).

### 2.5.5 Autocorrelation

An understanding of the autocorrelation (or serial dependence) in a system is important to the overall monitoring design and is especially significant for water quality monitoring. Autocorrelation in water chemistry results from lag effects or time delays due to a number of factors including physical characteristics, such as mixing and flushing rates, and chemical interactions, such as buffering capacities. For example, in a poorly buffered aquatic system (low alkalinity), pH fluctuations can vary significantly over short time intervals (low autocorrelation), while well buffered aquatic systems will exhibit small changes in pH over time (high autocorrelation).

An understanding of the autocorrelation in a waterbody is required to identify the optimal sampling frequency (both statistically and economically) when monitoring for mean concentrations. The estimation of the mean using the fewest samples requires that the sampling frequency be long enough such that each sample is independent of the previous one. When autocorrelation is high, sampling frequency requirements are low. In systems exhibiting a high level of autocorrelation, such as groundwater, parameter values are slow to change, and the optimum frequency of sampling is low. The situation is reversed when monitoring a small stream with high flow and low retention.

---

[4] Completed during the extensive survey or as part of the intensive baseline survey.

## 2.6 Quality Assurance and Quality Control (QA/QC)

### 2.6.1 General Principles and Protocols

An important component of any monitoring program is a comprehensive Quality Assurance and Quality Control program for both field and laboratory procedures. This section will only discuss the general principles of QA/QC procedures. QA/QC protocols applicable to specific sample matrices are addressed in the relevant sections.

***Quality Assurance***

The overall objective of Quality Assurance guidelines are to:

- optimize the generation of quality analytical data;
- ensure the documentation and defensibility of methods and data;
- ensure compatibility of methods and results from different proponents and contract services; and
- clearly outline the various QA/QC requirements that are required.

Before a monitoring program is implemented, the study design should be reviewed and approved by the operator, regulatory authorities, and relevant experts (e.g., a biostatistician). The review should evaluate:

- the extant to which the design meets the study objectives and uses appropriate tests of hypotheses;
- the technical elements of the program;
- the statistical elements of the design (including statistical power);
- economic costs (both the cost of sampling and the cost of the outcome); and
- the overall feasibility of the proposed design (e.g., technical, logistic, statistical, economic), considering appropriate modifications or possible alternate designs that would reflect the study objectives.

***Quality Control***

Quality Control (QC) commences with the collection and handling of samples in the field and delivery to the laboratory. A QCprogram should include the following requirements:

- appropriate education and/or training for personnel;

- appropriate sampling methodology (e.g., approach benthic invertebrate sites from downstream to minimize disturbance);
- the appropriate equipment for particular habitats (e.g., Hess cylinder for riffles and an Ekman dredge for depositional zones);
- consistent sampling methods throughout the study;
- proper maintenance of equipment, including documentation of the service and maintenance record;
- appropriate labelling for all samples;
- appropriate preservative/fixative;
- detailed field notes, in bound, paginated, water-proof notebooks;
- use by personnel of chain-of-custody forms and custody seals; and
- appropriate and safe shipping and storage methods.

Chemical analytical procedures should be completed in certified (e.g., CAEAL, ISO 9000, and ISO 14000) laboratories employing the following standard laboratory procedures:

- standard procedures for cleanliness applied to glassware, reagents, solvents, gases, instruments, and work surfaces;
- use of high quality reagents, checked for purity, strength, deterioration and contamination; and
- instruments routinely calibrated and maintained along with appropriate standards, with these procedures and results stored in operational logs.

The analytical procedure should also incorporate the analysis of Quality Assurance samples, shown in Table II:2.6-1, in order to define the precision and accuracy of the method for the sample matrix being analyzed. The appropriate method(s) will vary depending on the material being analyzed. There are also a number of field initiated Quality Assurance procedures that may be used to support the chemical analytical procedures. These are also detailed in Table II:2.6-1.

## 2.7 Data Management

The integrity of a monitoring program can be compromised if proper data management protocols are not initiated in the design stage. Monitoring programs serve little or no purpose if the data collected are not transcribed accurately and stored in a manner amenable for future analyses. Data should be stored in digital and hard copy formats with copies of field notes archived at a second location. Field books or notes should be reviewed prior to returning from the field to confirm that all of the required data have been recorded and are legible.

The screening of data should commence prior to transcription to additional or hard copy formats. Upon receipt of the results from the field or laboratory a preliminary review of the data should be completed paying attention to:

- anomalous values compared to previous sampling results;
- anomalies related to duplicates or test blanks;
- field and laboratory measurements which do not correspond, e.g., pH and conductivity; and
- values exceeding Tiering Thresholds and Action Limits[5].

Anomalous results should be reviewed immediately with samples re-submitted for analyses when possible. If sampling errors are suspected, sample protocols should be reviewed and quality assurance methods improved for the next sampling event.

Should tiering thresholds (triggers) be exceeded then the next tier should be implemented. This may involve analyzing the expanded parameter list on the collected samples or preparations for additional sampling in the next sampling period, depending on the site-specific study design.

Action Limits should be concentrations that are below those required for compliance in order to ensure that license requirements are not exceeded. Contingency quick-response-plans to be initiated when Action Limits are exceeded should be part of the overall environmental plan.

The preliminary review of the data should be completed by personnel familiar with the program and not by data entry personnel. Any corrections or changes should be made to the originals and to any copies with all changes initialed by a supervisor. After this preliminary review the data should be transcribed and proofed for accuracy. Data should be coded so that the the variables measured at a specific station at a specific time can be identified. Coding must be standardized with any changes to the coding system flagged and clearly documented. Values below limits of detection (LOD) should be stored/recorded as the detection value and coded to indicate they were less than the LOD.

---

[5] Action Limits are contaminant concentrations which are deemed unacceptable for release.

Decisions regarding the handling of these values (e.g. LOD, half the LOD, statistical calculation of the tail based on the frequency distribution, etc.) should be determined by the personnel completing the statistical analyses.

Additional data handling procedures are addressed in the following section on reviews and audits. Data handling concerns specific to certain sample components are addressed in the following sections.

## 2.8 Reviews and Audits

The monitoring program should be reviewed on a regular basis to ensure the program objectives are being met. At the start of a new sampling cycle, data from previous cycles should be reviewed to determine whether any additional tiering in the program should be implemented in the new sampling period.

The monitoring program should be reviewed in preparation for regulatory reporting periods and on a yearly basis as a component of the annual reports. This yearly assessment should address any difficulties encountered in the operation of the program, whether logistical, analytical, or procedural. An evaluation of the monitoring program over the full lifespan of the mine should be completed periodically (possibly every five years, e.g., Saskatchewan uranium mines Status of the Environment Documents). These reviews should incorporate:

- an analysis of the results to identify any impacts with special emphasis on detecting any trends in the data suggesting chronic impacts not readily apparent over shorter time scales;
- a comparison of observed versus predicted impacts to assess and improve the functioning of any modelling techniques used in the original EIA or the monitoring program;
- a summary of any “upset events” and assessment of the success of any control structures or remedial actions implemented;
- identification of any additional monitoring requirements or improvements, e.g., ensuring that the protocols are meeting the objectives and developing new monitoring techniques which are of value to the program;
- identification of redundant or unnecessary components that should be dropped from the program; and
- an assessment of whether the program is operating in a cost effective manner.

## 2.9 Section Summary

### Monitoring Framework / Design

There are three basic developmental stages of a mining operation:, the **pre-operational phase**, the **operational phase**, and the **decommissioning phase**.

During the pre-operational phase the **final mine plan** is completed, the **operational monitoring program is designed** and the **intensive baseline** is collected to support the operational monitoring program.

The development of the final mine plan involves two lines of investigation: project characterization and preliminary environmental characterization. **Project characterization** identifies the physical and structural components of the project which have the potential to impact the environment. **Preliminary environmental characterization** involves characterization of the ecosystems (abiotic and biotic components) associated with the mining project. The results of the project and environmental characterization programs are combined to identify potential impact pathways and complete impact/risk assessments. This information is used to identify operating and waste management procedures which minimize associated environmental impacts in the most cost efficient manner.

**Operational monitoring program** development commences with the classification of the monitoring requirements for each of the potential impact pathways. The monitoring classifications are described below.

***Compliance Monitoring*** is routine monitoring dictated by regulatory authorities as a condition for permission to develop, operate, or decommission a site.

***Diagnostic Monitoring*** is routine monitoring generally completed on the initiative of the operator focusing on potential AMD sources upstream of the compliance stations, thereby acting as an early warning system.

***Investigative Monitoring*** is usually initiated by the operator or required by regulatory agencies in response to non-compliance results, upset events (e.g. spills), or operational design changes.

An efficient overall monitoring framework incorporates monitoring **tiers** where core (Tier I) monitoring involves a suite of simple tests, with tests increasing in complexity and costs only if early results identify a need for more information. Compliance monitoring components / parameters are automatically classified as Tier I. The remaining identified diagnostic and investigative requirements are classified as Tier I, II, or III depending on an assessment of the trade-offs between their value as early warning indicators, and the expense associated with the monitoring component/parameter.

Once the components and tiering framework have been identified specific study designs must be developed for each component. Most source monitoring involves **sentry study designs**. These serve as early warning systems to prevent violation of licence requirements at downstream compliance stations.

Receiving environment monitoring usually requires **impact study designs**. The objective is to determine whether a measurable impact has occurred in the receiving environment as a result of mining activities and the monitoring involves one of four basic designs:

1) Control-Impact (**CI**) or Spatial Design;
2) Before-After (**BA**) or Temporal Design;
3) Before-After-Control-Impact (**BACI**) or Spatio-Temporal Designs; and
4) Gradient Designs.

The **intensive baseline** is completed only **after** the basic study design of the operational monitoring program has been determined. The intensive baseline should be treated as the first cycle of the monitoring program with sampling completed for all Tier I components/parameters. In addition, all identified Tier II and potential Tier III components/parameters that require baseline data should be collected.

During the **operational phase** the monitoring program is initiated as designed. Periodic reviews should be completed to ensure that the monitoring program is meeting the operational and regulatory needs of the project.

A **pre-decommissioning baseline** serves the same purpose as the pre-operational baseline. The objective is to classify and characterize the state of the potential AMD sources and their respective receiving environments immediately prior to, or after, closure. Impacted sites should be identified and characterized to monitor for recovery during the decommissioning phase. Monitoring will cease when sufficient data are available to support the conclusion that the decommissioning criteria have been met.

### Statistical Considerations

Determination of study design components such as replication requirements, sampling frequency, and the most appropriate statistical analytical methods should be based on the statistical characteristics of the preliminary and intensive baseline data. This requires identifying the power to detect a defined effect level, identifying the need for a stratified sampling design and determining the serial dependence (autocorrrelation) of the dataset.

### Quality Assurance and Quality Control (QA/QC)

A quality management system is important for assessing the validity of the dataset for both field and laboratory components. **Quality Assurance** guidelines are concerned with ensuring the adequacy of the study design as well as the generation, documentation, and defensibility of quality analytical data. **Quality Control** commences with the collection and handling of samples in the field, the delivery to the laboratory, and laboratory processing and analysis.

### Data Management, Reviews and Audits

The integrity of a monitoring program can be compromised if proper data management protocols are not initiated at the design stage. The screening of data should commence prior to transcription to additional or hard copy formats. Upon receipt of the results from the field or laboratory a preliminary review of the data should be completed.

The monitoring program should be reviewed on a regular basis to ensure the program objectives are being met. Reviews should be completed at the start of a new sampling cycle, in preparation for regulatory reporting periods, and on a yearly basis as a component of the annual reports. An evaluation of the monitoring program over the full lifespan of the mine should be completed periodically (e.g. 3 to 5 yrs).

## 2.10 Section Bibliography

Theoretical Study Designs
Eberhardt and Thomas 1991
Green 1979
Underwood 1991
Underwood 1992
Underwood 1994

Study Design Statistical Concepts
Bajpai et al. 1992
Gilbert 1987
Green 1979, 1984, 1989, 1993
Stewart et al. 1986
Winer 1971

Sample Size and Power Analysis
Alldredge 1987
Bernstein and Zalinski 1983
Goldstein 1989
O’Brien and Muller 1993
Peterman 1990
Mapstone 1995

Monitoring Theory
Chapman et al. 1982
Skalaski and McKenzie 1982
Valiela and Whitfield 1989
Ward 1986
Karr 1987

Risk Assessment
Clifford et al. 1995
Lyne et al. 1992
Suter 1993. 1996
Suter and Loar 1992

Quality Assurance / Quality Control
BC AMD 1992
Cuffney et al. 1993
Gilbert and Kinnison 1981
Liabastre et al. 1992
U.S. EPA. 1980, 1986a, 1987a, 1991a

Statistical Texts and Papers
Hoke et al. 1990
Hopke 1992
Bajpai et al. 1992
Conover 1980
Davison 1992
Day and Quinn 1989
Draper and Smith 1981
Gilbert 1987
Legendre and Fortin 1989
Snedecor and Cochrane 1989
Sokal and Rohlf 1981
Tabachnik and Fidell 1983

Data Handling / Detection Limits
Adams 1992
Balasundaram and Shashidhara 1993
Bowen et al. 1991
Grubbs 1969
Porter 1986
Gilbert and Kinnison 1981

Tiering
Hodson et al. 1996
Keough and Mapstone 1997

Audits and Reviews
Buckley 1991
Bowen et al.

# CHAPTER III - SOURCE EVALUATION AND MONITORING

The first step in developing a pro-active source monitoring program is to evaluate the AMD potential for the range of possible AMD sources. Information on AMD source evaluation is provided in Section 1.0 of this chapter. The results of the AMD site evaluation, are then used to identify and develop monitoring requirements specific to the site. This process is addressed in Section 2.0 of this chapter.

## 1.0 PREDICTION OF AMD POTENTIAL

Characterization and possibly predictive modelling of a potential AMD source are required for the following reasons:

- to predict future effluent water quality for operational or decommissioning planning;
- to evaluate decommissioning options;
- to evaluate effectiveness of a cover over waste rock or tailings;
- as a licensing or closure planning requirement;
- as justification for the amount of a financial assurance mechanism (i.e., letter of credit or secure bond);
- to calculate long-term treatment costs; and
- to provide modelling estimates of potential inputs to the downstream environment.

Historically, AMD management has focussed on control, and/or collection and treatment after acid generation has occurred. Control, whether through inhibition of acid generation or through containment (e.g., covers and seals), has proven to be difficult once acid generation has commenced. The collection and treatment of AMD, though costly, has usually been the only reliable solution.

Recently, the focus has been to anticipate the formation, extent, and impact of AMD before mining and milling commence. Accurate prediction of AMD is necessary to develop a cost-effective means of reducing the generation and impact of AMD. Predictive assessments allow advanced planning of waste management and the incorporation of control and treatment procedures in the Final Mine Plan. In addition, consideration of potential AMD impacts is now a prerequisite for the permitting or expansion of new mining and milling operations (BC AMD 1989) and generally falls under *due diligence requirements* of lenders. Hence, failing to incorporate predictive AMD procedures into a project's Final Mine Plan may result in expensive delays in the approval process and leave the operation vulnerable to unforeseen and unbudgeted expenses from future environmental AMD problems.

## 1.1 Approach to AMD Prediction

The prediction of AMD serves two objectives:

1) to provide sufficient data to satisfy the mine operator/owner, regulatory authorities, and the public that potential AMD concerns can be managed; and
2) to focus the AMD monitoring program in an efficient cost effective manner.

A number of approaches may be employed for estimating AMD potential. Several possible approaches are summarized below, and additional information is provided in Table III:1.1-1:

- previous experience with the mine waste and geological or geographical comparisons to nearby minesites where AMD potential has already been expressed and evaluated;
- paleoenvironmental and geological models;
- chemical, mineralogical, and physical analyses of the waste;
- testing to identify the readily extractable metals;
- static prediction tests for quantification of the balance between acid producing and acid consuming components of the waste;
- kinetic prediction tests for determination of the rates of mineral dissolution, acid generation, and metal release, in both laboratory and field; and
- predictive modelling of potential future AMD production.

The framework for the development of an AMD Prediction Program is outlined in Figure III: 1.1-1. MEND Report 1.16.1a and 1b (1989 and 1991), as well as BC AMD (1989), provide a great deal of information on developing and completing predictive AMD programs. The suggested approach is to subdivide the AMD Prediction Program into three components: Exploration, AMD Assessment (Stages I/II), and AMD Mine Plan (Stage III). These three components are discussed briefly in the following sections.

### 1.1.1 Exploration

The objective is to provide a **preliminary** assessment of acid generating potential. The geologist should complete the following procedures:

- map the geology of the ore deposit and identify the exploitable ore zones;
- prepare a mineralogical assessment of prevalent rock types; and
- provide a weathering assessment of natural outcrops and identification of secondary minerals, if present.

MEND 1.32.1 (1993) proposed that this should involve collection of the following field measurements/observations by geological and environmental personnel.

| Sample Material | Program Stage (See Figure II:2.2-1) Responsibility | Measurements/Observations |
|---|---|---|
| Rock<br>Adapted from MEND 1.32.1 by Kwong 1993 | Project Characterization<br>Geologist | Lithology and moisture content (qualitative)<br>Mineralogy and texture<br>Nature and extent of weathering<br>Fracture/joint density; foliation geometry |
| Weather Products<br>e.g. gossams and soils | Project Characterization<br>Geologist | Mineralogy and texture<br>Colour and moisture<br>Profile development<br>Paste pH<br>Vegetation, if any (Basic assessment) |
| Waterbodies | Environmental Characterization<br>Biologist/Ecologist | pH, Eh and conductivity<br>Acidity, alkalinity and sulphate, if possible<br>Presence of secondary precipitates<br>Evidence of microbial activity (e.g. biofilm) |
| Others | Environmental Characterization<br>Biologist/Ecologist | Hydrometeorology (climate)<br>Landscape<br>Vegetation |

The geologist should also be consulted to help identify and map the existing ARD source(s), if acidified drainage waters are identified during the completion of the environmental characterization (see ChII:2.2.1.1 and ChIV:2.1.3.2).

### 1.1.2 Environmental Evaluation

***Preliminary (Stage I)***

A review of the available AMD information from other mines with similar geographical, geological, and paleoenvironmental characteristics should be completed. This information can be used to refine the following steps in the AMD Predictive Program.

Further delineation of the orebody (e.g., extensive drilling, trenching, and exploration adits) may provide additional material for a more detailed assessment of the potential rate of oxidation of **subsurface** materials.

The project geologist should supervise a detailed **identification** of geological subunits of the ore and waste rock based on the preliminary assessment, which should include the following minimum steps:

- core logging to identify mineralogy, porosity/permeability, and structural integrity;
- petrographic analysis to obtain the identity and texture of the sulphides present and the associated gangue minerals; and
- simple, rapid field tests such as the fizz and paste pH/conductivity tests.

**Representative** samples (see ChIII:1.4) should be collected from each geological unit for **physical**, **chemical**, and **mineralogical** analyses, and **static tests** to estimate the potential for net acid production (see ChIII:1.2). The next step is to examine the chemical and mineralogical analyses, and the static test results, and evaluate the **homogeneity** of the geological subunits. Statistical methods can be employed to test for homogeneity. If a specific lithologic unit has highly variable results, then the unit should be subdivided into additional subunits and re-sampled.

If the static tests identify any lithologic unit(s) as potentially acid generating, then additional detailed characterization is required in Stage II. If some of the units were identified as non-acid generating, then the testwork suggested in Stage II may not be necessary, and one can proceed to Stage III.

***Detailed (Stage II)***

The following procedures should be completed if the static test results identify a specific lithological unit(s) as being potentially acid generating:

- collect additional representative samples (see ChIII:1.4) from each lithological unit identified by the static tests as potentially acid generating;
- perform **kinetic testwork** (see ChIII:1.3) to confirm acid generating potential and to estimate rates of sulphide oxidation, acid generation, neutralization, and metal depletion;
- commence development of waste management plan(s) incorporating AMD prevention and control procedures;
- complete hardness and weathering assessments to support predictive modelling; and
- apply predictive methods and/or mathematical modelling to extrapolate long-term acid generation and possible water quality trends from the static and kinetic test data.

If the available data are deemed sufficient to support the predictive modelling, then the investigation moves to Stage III; otherwise, additional samples or alternate testwork may be required to support further interpretation and predictive modelling.

### 1.1.3 AMD Mine Plan (Stage III)

Once the preliminary characterization (Stage I) and testwork (Stage II) have been completed, a detailed waste management plan should be developed to address any potential AMD concerns. If AMD testwork was negative (no AMD potential) there should then be a final review of the test results to determine whether further testing is justified. The final monitoring plan may now be completed, incorporating the results from the Environmental Characterization (Stage I) investigations, with consideration of potential impacts from both acid-generating and non-acid generating materials (see Figure II:2.2-1).

## 1.2 Prediction Methods

### 1.2.1 Static Tests

Static tests are rapid, relatively inexpensive procedures which measure the net acid potential (NAP) of a sample by defining the balance between acid generating ability (AP - Acid Potential) and acid neutralizing capability (NP - Neutralization Potential). Theoretically, a sample will generate net acidity only if the acidity generated exceeds the neutralization capacity of the material; i.e., NAP = AP - NP is positive. An alternate approach is to consider the net neutralization potential (NNP = NP - AP), in which case a net acid generation capacity is indicated by a negative NNP value. This practice is commonly referred to as acid-base accounting (ABA).

There are several types of static tests for determination of NP such as the Sobek[6] method, the Modified Sobek Method, and the BC Research Initial Test (Table III:1.2-1). The paste pH and fizz tests are simple field tests that can provide a quick indication of the net free acidity/alkalinity and the presence of carbonate buffering. Additional information on static test options is provided in Table III:1.2-1 and TSN (2.1).

Static tests cannot be used to predict the future quality of drainage from waste materials since acid generation and drainage quality are time dependent functions incorporating a number of complex factors including mineralogy, rock structure, and climate. Hence, interpretation of static tests is limited to identifying the **potential** for net acid generation at some unknown point in time. Such interpretation typically involves application of criteria developed from general mining experience to ABA parameters such as the NAP, NNP, and the NP/AP ratio. For example, an NP/AP ratio less than 1, or an NNP value less than -20, is typically assumed to indicate high potential for acid generation.

### 1.2.2 Kinetic Tests

The static tests will identify the geological units which display some potential for acid generation. Kinetic tests are completed to provide information on the possible rates of acid generation, and the time-frame involved. This information is necessary to address the specific severity and possible duration of acid drainage and to support the selection of the prevention/control/treatment processes that will be required to minimize the overall costs for the abatement of acidic drainage.

Selection and design of kinetic testwork are more complex than static testwork investigations. The decisions and procedures involved in a kinetic test program are described below and a schematic is provided in Figure III:1.2-1.

#### *Sample Material Characteristics*

The initial step is to identify the material characteristics of the sample including:

- particle size;
- mineralogy;
- total metal analysis; and

---

[6] Also known as the U.S. EPA Method.

- sulphur and carbonate content (from static tests).

### *Define Program Objectives*

Selection of a specific kinetic test will depend on the program objectives and may include one or more of the following objectives:

- determination of the possible reaction rates for acid generation;
- examination of the influence of specific environmental factors on the rates of acid generation/neutralization, and the release of contaminants;
- investigation of the buffering minerals, such as dissolution rates and availability for neutralization reactions;
- identification of the major contaminants of concern and the possible concentrations of these species in the seepage;
- determination of maximum concentrations of contaminants and possibility of exceeding water quality objectives;
- determination of the effect of flushing rates on water quality;
- determination of the influence of bacteria on the acid generation; and/or
- selection or confirmation of disposal option(s).

### *Select Kinetic Test Procedure*

A brief summary of the objectives, advantages, and disadvantages of the most common kinetic test procedures is shown in Table III:1.2-2. Additional information is provided in TSN 2.2.

### *Interpretation of Kinetic Test Results*

The primary objective of kinetic testwork is to predict both the rate and extent of acid generation. The various types of kinetic testwork will provide useful information; however, at present there are no standard procedures for interpretation of the results. It will be difficult to translate the results obtained from the kinetic testwork directly to the field situation.

The kinetic testwork results should be assessed to determine if the proposed Mine Plan will be environmentally acceptable. If the test results indicate a potentially unacceptable environmental situation, then the Mine Plan, should be refined. If the existing test results do not support the refined

Mine Plan then additional testwork will be required. Once adequate information has been obtained from the testwork, the Mine Plan can be examined in greater detail using predictive modelling.

### 1.2.3 Modelling

Mathematical models are often used to extrapolate the results from both the static and kinetic testwork into long-term predictions of water quality. Model selection may range from simple curve fitting exercises to the use of sophisticated computer models of the processes governing acid generation. Additional information on modelling is provided in the documents listed in the bibliography.

Different prediction methods are usually best suited for application at specific points in the sampling and monitoring program. For example, examination of ABA parameters could be performed on the initial and subsequent samples to characterize and rank the potential for acid generation. The results of kinetic testwork would not be available until later in the program, but these could be interpreted through comparison to field monitoring data and through application of statistical methods to obtain long-term predictions (i.e. extrapolations) from the testwork data. As the number of samples in the monitoring database increases, statistical methods could also be used to evaluate possible trends in water quality. Chemical speciation models such as MINTEQ can also be applied to determine the influence of specific minerals on the water quality.

Predictive (computer) models use mathematical relationships to represent or simulate the important physical, chemical, geochemical, and biological processes that control acid generation. The prediction of acidic drainage is rather complex, as these important processes are inter-related, and vary with time. Some mathematical models have been developed to examine selected processes in greater detail (e.g. physical), while others have been developed to examine the combination of these processes.

Predictive modelling can be performed to support the following objectives:

- obtain a better understanding of the mechanisms of acid generation;
- examine the current extent of acid generation;
- predict the future contaminant loads and concentration profiles;
- assess future treatment requirements;
- compare management or decommissioning options;
- characterize and predict future contaminant transport; and

- predict possible violation of permit criteria and potential for environmental impacts.

Several factors may influence a model's performance under field conditions. Adequate data (field and laboratory) must be available for calibrating and validating the computer model. This will improve the confidence in the long-term predictions, which may be made several hundred years into the future.

## 1.3 Sampling for Static and Kinetic Tests

### 1.3.1 Sampling Locations

A number of mine components require representation in the sampling program. The sampling locations and requirements will differ depending on whether the mine is in pre-operational or operational stages of development. The potential sources of samples for the most common mine components requiring predictive AMD assessment are provided in the table below.

| Mine Component | Existing Mine | Proposed Mine |
|---|---|---|
| Pit walls | Drill core<br>Pit walls | Drill core<br>Underground exploration passages<br>Trenches |
| Underground workings | Drill core<br>Walls<br>Excavated rock | Drill core<br>Underground exploration passages |
| Waste rock / overburden piles | Waste rock piles<br>Drill core | Drill core<br>Underground exploration passages |
| Tailings | Tailings<br>Impoundments | Pilot plant for mill process |
| Ore stockpiles | Ore stockpiles | Drill core<br>Underground exploration passages |
| Spent ore | Heap leach | Pilot plant for heap leach |

Reproduced from BC AMD 1989.

The geologist can identify the various lithological units that should be defined on the basis of a preliminary assessment of their acid generating potential (e.g. based on mineralogy). Samples should be collected from rock units that will be exposed during excavation and mining, accounting for the mining sequence and the quantities of each unit that will need to be managed.

### 1.3.2 Sampling Considerations

The objective is to collect a sufficient number of representative samples to obtain the desired characterization of the mine wastes. Sampling strategies should be site-specific, but there are common elements that constitute an appropriate sample design. A staged or stratified approach where the number of samples collected from a geological subunit is related to the relative contribution of that unit to the total ore or waste rock removed is recommended. Analyses of the variation in the ABA of the initial samples will provide guidance as to how many more samples may be required and which geologic units should be re-sampled.

For example, a curve relating the minimum number of samples to be collected from each subunit, based on the mass of the unit, is provided in Figure III:1.3-1 (BC AMD 1989). This curve was developed from unpublished field data, but the sample numbers obtained from this relation are reasonably consistent with industry practice. An approximate mathematical expression of this curve

$$N = 0.026\ M^{0.5}$$

is:

where: N = number of samples
M = mass of the geological unit in tonnes ($M > 6 \times 10^3$ tonnes) (MEND 1994)

Due to the preliminary nature of this curve, it should be used only as a guide and application should be limited to highly homogenous units.

## 1.4 Data Management

The data obtained from the AMD assessment program should be stored in a manner conducive to periodic review and refinement of the program. One of a large number of commercially available spreadsheet or database computer applications could be tailored into a site-specific database. A site-specific database structure for personal computers could also be developed based on the spreadsheets proposed by Smith and Brady (1990) for evaluating acid base accounting data.

## 1.5 Section Summary

Evaluating the AMD potential of a minesite is essential to identify monitoring requirements. An AMD prediction program is completed during the pre-operational phase of a mine and is continuously re-assessed and fine-tuned during the operational phase. The pre-operational program can be subdivided into three components:

1) Preliminary Exploration Assessment
2) AMD Assessment, and
3) AMD Final Mine Plan.

**Exploration assessment** is a preliminary determination of the acid generating potential of the site by the site geologists. Exposed materials, whether natural outcrops or mechanically exposed (e.g. exploration trenches), are assessed and drainage from these areas is monitored.

Formal **AMD assessment** commences with a review of the available AMD information from other mines with similar geographical, geological, and paleoenvironmental characteristics as identified in the exploration assessment. The data from the preliminary assessment also serve to identify geological subunits of the ore and waste rock for **stage I AMD assessment**. This involves identifying simple physical and mineralogical characteristics, completing rapid acid field tests and, when acid potential is indicated, static acid base accounting.

**Stage II AMD assessment** is completed on lithological unit(s) identified in Stage I as potentially acid generating. **Kinetic** acid generation test work is completed to confirm acid generating potential and to estimate **rates** of sulphide oxidation, acid generation, neutralization, and metal depletion. Additional hardness and weathering assessments are also completed to support **predictive modelling** for extrapolation of long-term acid generation and possible water quality trends from the static and kinetic test data.

With confirmation of acid generating potential, detailed **AMD waste management procedures** are developed for the mine site and incorporated into the **Final Mine Plan**. The final monitoring plan may now be completed, incorporating the results from the Environmental Characterization investigations, with consideration of potential AMD impacts.

## 1.6 Section Bibliography

**Guidelines for Predictive Assessment**

BC AMD 1989, 1990a
Byerly 1990
Higgs et al. 1997
Hutchison and Ellison 1992
MEND 1.16.1a,b 1991
MEND 4.5.1-1 1994
Miller 1996
Price et al. 1997a,b

**Assessment: Exploration Phase**

MEND 1.32.1 1992
Kwong and Whitley 1992

Miller et al. 1991b**Sampling**
BC AMD 1989.
MEND 4.1.1 1989
MEND 4.5.1-1 1994
MEND 4.5.1-2 1994
NUTP 1985

**AMD Predictive Modelling**

BC AMD 1990a
MEND 1.42.1 199?
Hart et al. 1991
Lin et al. 1997

**Test Work**

Day et al. 1997
Kwong and Ferguson 1997
Lawrence and Wang 1997
MEND 1.16.1a 1989
MEND 1.16.1b,c 1991
MEND 1.16.3 1996

**Regulations and Legislation**

Errington 1991
Lindahl 1991
MEND 4.2.1 1993
Price et al. 1997a

## 2.0 ROUTINE SOURCE MONITORING

This section addresses **routine** monitoring of various sources (e.g. waste rock piles, tailings management areas, mine workings such as pit walls and pit floors, and releases from AMD treatment facilities). Standard compliance and diagnostic source monitoring is concerned with:

- confirming that the mine is meeting contaminant compliance regulations (**compliance**);
- serving as an early warning indicator of possible violation of compliance regulations (**diagnostic**); and
- providing early warnings of commencement of, or an increase in, acid generation (**diagnostic**).

These objectives are primarily achieved by monitoring the chemical composition of surface water and groundwater prior to, or at the point of entry into the receiving environment. Tier I source monitoring will primarily consist of chemical analyses of surface and ground waters using a Tier I parameter list. Tier I parameters will consist of parameters required by a permit at the compliance stations and parameters which serve as early indicators of acid generation.

Non-compliance results or evidence of the commencement of, or an increase in, acid generation and or trace metal concentrations, will result in the initiation of Tier II monitoring. The Tier II objective is to confirm the effect detected in Tier I, and identify whether the effect persists. In most situations this can be achieved by initiating more comprehensive surface and groundwater chemical analyses by testing for the selected Tier II analytical parameters.

Once the effect has been identified as persistent or increasing, then Tier III, consisting of **investigative** monitoring components and parameters, will be initiated. Monitoring components will extend beyond the basic components of surface and ground water as the program shifts into Tiers II and III. The exact components will be source (e.g., waste rock pile, tailings management facility) and mine site-specific. These are addressed in greater detail in the Technical Summary Notes (TSN). The focus in the following sections is on **routine** operational and decommissioning source monitoring requirements common to most mine sites.

The importance of a pre-operational assessment of the hydrology and quality of both surface and groundwater is recognized, and addressed in ChIV:1.0 and 2.0. This section should also be consulted for additional supportive information (e.g. QA/QC, sampling protocols, etc.) on conducting hydrological and water quality investigations.

## 2.1 Classifying Priorities for Pathways and Stations

In the following sections, examples of generic contaminant pathways are detailed for each of the above mentioned source types. Recommendations are outlined for locating, along these generic flowpaths, monitoring components and stations.

Potential contaminant pathways differ in their priorities from a monitoring and risk assessment perspective. Compliance stations are located on priority pathways and are closely followed in importance by diagnostic pathways/stations in the routine operation of a minesite. Investigative monitoring components, while not necessarily a priority in the routine operation of a mine, may become significant under certain site-specific conditions.

### Compliance monitoring paths/stations (C)

Stations located at points of compliance as designated by operating permits.

### Primary diagnostic monitoring paths/stations (PD)

Routine high priority diagnostic monitoring pathways/stations. These are diagnostic stations which act as early warning sites and should be common to most mining sites. They would generally be located at:

- the beginning of the pathway close to the source;
- points of discharge from an artificial (e.g., drainage ditch) to a natural water course (e.g., stream on or off property); and
- points where identified contaminated flowpath(s) interact
  - e.g., where shallow contaminated groundwater flows interact with surface flows.

### Secondary diagnostic monitoring paths/stations (SD)

Additional **diagnostic** monitoring stations which tend to be required on a site-specific basis. They are required:

- when AMD concern is high;
- when the pathway is long (between source and compliance station); and/or
- where multiple point sources combine
  - e.g., drainage ditch or on-site creek receiving input from different sources.

**Investigative monitoring paths/stations (SD)**

- Additional monitoring stations for **investigative** purposes generally restricted to sites with high AMD problems or concerns. These tend to cover special short-term investigations, employing new or emerging technologies or research.

## 2.2 Waste Rock

### 2.2.1 Background

The quantity and nature of the waste rock can vary widely, depending on the type of deposit and its location. Generally waste rock piles consist of a heterogeneous mixture of several rock types (lithologies), each with differing potential for acid generation. In addition, there may be zones of geochemical alteration within a rock type or extending across rock types which will alter the AMD potential. The various rock types may be segregated during disposal or, more commonly, mixed within a waste rock dump, with the distribution controlled by the order of excavation and the way the dump is constructed.

### 2.2.2 Pathways

A conceptual diagram of the acid generation process and AMD migration patterns through a representative waste rock pile is shown in Figure III:2.2-1. There are three potential sources of water for a rock pile:

1) groundwater;
2) surface water; and
3) precipitation.

The most significant input is likely to be precipitation, as a properly designed pad should eliminate or minimize non-precipitation inputs. Precipitation infiltrates through the pile to the pad where it may:

- mix with upwelling groundwater and flow from the basal perimeter of the pile;
- enter the underlying groundwater flow system; and/or
- exit at the basal perimeter of the pile.

If the pile is on a low permeability pad, interaction with the groundwater should be limited. These pathways can be further simplified into a flowpath schematic as shown in Figure III:2.2-2. This schematic should be customized according to site-specific considerations with the elimination of flowpaths not relevant to the site. In Figure III:2.2-2, portions of the flowpaths are coded according to the monitoring priority of the flowpaths (primarily applying to water quality monitoring).

### 2.2.3 Compliance and Diagnostic Monitoring

The principles of monitoring waste rock piles can be applied to other site-specific mine rock locations such as marginal ore stockpiles, heap leach piles, and "clean" waste rock fill used in parking lots and local roads. In general, **compliance monitoring** is usually not required within the immediate source area of the piles. Hence, most waste rock monitoring programs are **diagnostic** in nature with the primary focus on surface and groundwater quality. They are often conducted on local seepages and run-off from the piles with the objective of detecting changes in water quality and locating specific sources of concern before water quality at the designated point(s) of compliance becomes a concern.

Most waste rock pile **investigative monitoring** consists of re-assessment of AMD potential (to verify or compare to pre-operational predictions as discussed in ChIII:1.0) or the collection of data required for modelling purposes (Table III:2.2-1). Internal monitoring of rock piles tends to fall into the investigative monitoring categories (Figure III:2.2-2).

#### 2.2.3.1 Identifying Sampling Locations

***Primary Diagnostic and Secondary Diagnostic Stations***

*Surface Waters*

**Primary** surface water quality stations should be located at the following points:

- at the toe of waste rock piles or in the surrounding collection ditch; and
- at any points of discharge from the drainage ditch to a natural water course (e.g., stream on or off property).

If the waste rock pile does not mark the headwaters of the drainage channel, whether natural or constructed, then an upflow primary monitoring station (Control) should be established to delineate natural ARD contributions from waste rock AMD contributions.

To optimize AMD treatment, a **primary** or **secondary diagnostic** monitoring station, depending on the sensitivity of the treatment process, is required to analyze waste rock waters prior to their entry into a treatment system. This will likely involve:

- contaminated water collected from the toe of the pile or the surrounding drainage ditch (whether infiltrate or surface flow); or
- surfaced or intercepted groundwater requiring treatment.

Secondary diagnostic stations for surface waters would only be required when AMD concerns are high. They would be used to:

- delineate drainage from different potential AMD sources; and
- monitor between the primary diagnostic station and a compliance station when the distance between the two is such that significant changes in water quality may be expected.

*Groundwater*

**Primary** groundwater monitoring stations are required if there is the potential for seepage into the shallow groundwater system.

- where groundwater is a concern, samples should be collected
  - from daylights[7] located along the flowpath,
  - at well(s) sunk along the flowpath when groundwater contamination has been identified as a concern.

A minimum sampling program would involve one up-gradient and one down-gradient well for each hydraulically related stratum potentially impacted by the waste rock pile.

---

[7] Where groundwater flow appears at the surface.

Geophysical methods (e.g., ground and air electromagnetic surveys, TSN 8.0) are currently under investigation to determine their viability for monitoring the migration of AMD from waste rock pads and TMFs into the groundwater or through the overlying vadose zone. Additional information on groundwater monitoring is provided in ChIV:1.2 (hydrological requirements) and ChIV:2.2 (water quality), with information on specific sampling techniques provided in the TSN 14.0 and in MEND 4.6.1 (1994).

#### 2.2.3.2 Identifying Sampling Periods / Frequency

Waste rock piles possess a number of specific characteristics important to the determination of a suitable sampling frequency. As discussed in ChIII:3.3, AMD contaminants may accumulate within the rock pile, only to be released to the receiving environment as a pulse when heavy rainfalls or snow melts flush the pile. In cold climates, internally generated heat from chemical/biological reactions within the waste rock pile may result in water infiltration and flushing prior to run-off in surrounding areas. Increases in surface flow paths when shallow groundwater paths are still frozen should also be accounted for. Hence, fixed frequency sampling (BC AMD 1989 recommends monthly) will poorly represent monthly contaminant discharges and may completely miss peak discharges. At a minimum, monthly sampling programs should be supplemented by additional sampling immediately after (or during, depending on storm duration) heavy rainfall events, or planned to coincide with rapid snow melts. A more detailed discussion on the appropriate statistical approach to developing a seasonally stratified sampling protocol for water quality source monitoring is provided in ChIII:2.5.

### 2.2.4 Investigative Monitoring Components

Most internal monitoring of rock piles would be classified as investigative monitoring due to the complex dynamics and heterogeneity of acid generation within rock piles. Investigative monitoring typically supports additional modelling to re-assess predictive AMD potential. Investigative monitoring may involve:

- determination of the rates and pathways for internal water movement,
  - e.g., identifying saturated, discrete channel flow, and/or lateral flowpaths;
- measurements of water infiltration rates to develop water balance estimates (TSN 3.2);
- determination of water content (TSN 3.7) and porosity (TSN 3.5);
- pore water collection for chemical analyses (TSN 14.2.1, 14.2.2);

- use of peizometers for water sampling if a water table is present within the rock pile (TSN 14.3);
- gas sampling to measure by-products of acid production such as ($CO_2$) and measurements of oxygen availability and diffusion rates (predictive modelling or cover assessment, TSN 3.2) via gas sampling (TSN 3.1), and permeability (TSN 3.4.2) or porosity (TSN 3.5) measurements;
- use of temperature probes to detect temperature fluxes resulting from chemical reactions within the source or to assess bacterial development rates (TSN 3.6);
- monitoring bacterial populations (TSN 3.7); and
- deep groundwater monitoring (tends to be categorized as investigational due to the expense associated with the sinking of the wells) (TSN 14.3).

## 2.3 Tailings Management Facility (TMF)

### 2.3.1 Background

A TMF is an engineered facility for controlling, securing, and permanently disposing of tailings solids. An impoundment is constructed by combining dam or dyke walls with natural topography or by using mined-out open pits. TMFs range from simple impoundments constructed from clean waste rock and overburden (Figure III:2.3-1) to more complex structures incorporating permeable liners and other design enhancements (Figure III:2.3-2).

Tailings are discharged into the constructed impoundments using one of three deposition methods which can influence the monitoring requirements:

- drained or sub-aerial “slurry” deposition (Figure III:2.3-2, Stage I)
  - spigotting of the slurry from a pipeline with surface water decanted from the tailings solids and re-cycled in the milling process;
- subaqueous deposition (Figure III:2.3-2, Stage II)
  - tailings are discharged under water, with a pond or supernatant maintained over the tailings solids; or
- dry deposition
  - tailings are handled in a solid form.

AMD generation in tailings differs from rock piles in a number of ways:

- the fine particle size of tailings results in a more homogenous distribution of potentially acid-generating sulphide minerals and neutralizing-alkali minerals; hence, there is a more homogenous acid/neutralizing balance relative to rock piles;
- the fine particle size also results in a lower permeability restricting oxygen penetration and the rates of infiltration and migration;
- the mill effluent discharged as the water component of the tailings slurry tends to be strongly alkaline, neutralizing acid generation on previously exposed tailings; and
- when deposited as a slurry the tailings tend to be completely or partially saturated thereby limiting oxygen exposure and hence oxidation. As a result oxidation tends to be restricted to exposed surfaces along beaches or high ground areas of the TMF. The use of sub-aqueous deposition eliminates this exposure.

Tailings particles tend to be sorted when they are hydraulically placed, and sulphide minerals may be closely associated with a specific size fraction of the tailings. The content and particle size distribution of sulphide minerals within tailings are site-specific. The distribution of sulphide minerals within a specific TMF is an aspect that needs to be understood for the development of tailings solids, pore water, and pore gas monitoring programs. Other factors to consider when developing a sampling program at a TMF are:

- possible oxidation of near surface sulphide tailings
  - the depth of oxidation will vary from site to site and across a TMF;
- the possible presence of other materials in some areas of the TMF
  - some TMFs are used to dispose of waste rock and lime treatment sludges;
  - landfill areas may also be contained within TMFs;
- the nature of the tailings and the potential instability of the tailings surface; and
- the presence of impermeable liners in some tailings basins.

### 2.3.2 Pathways

A conceptual diagram of a simplified TMF and the potential acid-generating processes is provided in Figure III:2.3-1, with a pathways schematic provided in Figure III:2.3-3. For most TMFs there are two primary water sources:

1) precipitation; and
2) water in the tailings slurry.

Oxidation tends to be restricted to the upper or vadose zone of the tailings, but moves downward over time. High water levels in the tailings impoundment results in the downward and lateral movement of any AMD generated in the oxidation zone. This pore water tends to be neutralized as it moves deeper through the tailings, though the neutralizing capacity will decrease with time as the alkaline components of the tailings are consumed. This downward lateral movement can result in **embankment seepage** and **groundwater discharge**. Due to the low permeability of the tailings, it may take months or years for AMD generated at the surface to enter the groundwater.

TMFs designed with side drains (Figure III:2.3-2) collect the laterally moving pore water and direct it to a bottom drain from which it is pumped to a treatment facility (Figure III:2.3-3). Water ponding on the tailings surface can also be pumped from the TMF for treatment. The remaining potential contaminant pathway is **surface discharge** from a spillway. Such release would be rare as most TMF water is re-routed for treatment.

### 2.3.3 Compliance and Diagnostic Monitoring

**Compliance** monitoring should be completed at the stations and using the protocols as detailed in the permits for the mining operation. In most cases compliance monitoring will be restricted to surface water quality monitoring at the point(s) of release from the TMF to the defined receiving environment. For TMF facilities which interact with large groundwater systems, compliance monitoring may also involve a groundwater component.

**Diagnostic** monitoring of a TMF tends to be more intensive relative to the monitoring of waste rock piles due to the high contaminant loads and fine grained particles. Diagnostic monitoring of TMF's should incorporate water quality analyses of surface waters, drainage waters, groundwater, pore water, and tailings characteristics.

#### 2.3.3.1 Identifying Sampling Locations and Periods / Frequency

A schematic of the AMD pathways for a generic TMF is provided in Figure III:2.3-3 along with recommended monitoring priorities. The classification system is defined in ChIII:2.1 with the locations subdivided into Compliance, Primary and Secondary Stations. A list of the monitoring options and generic classification of their status (i.e., P, SD, SI) is provided in Table III:2.3-1.

***Primary and Secondary Diagnostic Stations***

*Surface Waters*

**Primary** surface water quality stations should be located at all points of release to the environment upflow of the compliance stations during the operational and decommissioning phases. This would involve:

- monitoring of embankment seepage waters;
- monitoring water directly released to the environment from the TMF; and/or
- end of pipe monitoring of water released from an associated AMD treatment facility.

This assumes that the latter two points have not already been designated as compliance stations by the regulatory bodies.

The chemical composition of the slurry (both tailings solids and liquid effluent) should be monitored prior to deposition.

In order to refine treatment protocols, a **primary** or **secondary diagnostic** monitoring station, depending on the sensitivity of the treatment process, is required to analyze water pumped to a treatment system. This may involve:

- water pumped from surface ponds formed in TMFs using sub-aerial deposition;
- water pumped from TMFs using subaqueous deposition or flooded decommissioned TMFs still being treated; and
- water pumped from TMFs with side and bottom drain systems (Figure III:2.3-2)
  - monitoring drain water is also an excellent means of assessing AMD production and downward seepage through the tailings.

In flooded TMFs, whether subaqueous deposition systems or flooded after decommissioning, *in situ* water quality should be assessed using a minimum of one deep water station on a monthly basis. If the TMF waters stratify, samples should be collected from the epilimnion, metalimnion and hypolimnion (see ChIV:2.0). If no stratification occurs, samples should be collected from the top, middle and bottom (ensuring no tailings or sediment cap contamination) of the water column. These samples should **not** be composited.

*Groundwater*

**Primary** groundwater monitoring stations are required if aquifers intersect with the TMF. A minimum sampling program would involve one up-gradient and one down-gradient well for each hydraulically related stratum potentially impacted by the TMF (BC AMD 1989). The higher the hydraulic conductivity of the subsurface strata the more dense the groundwater network (additional downflow stations) should be and the greater the sampling frequency.

The importance of the groundwater monitoring network increases during the later operational and decommissioning phases as water slowly migrates downward through the low permeable tailings. Hence, it may initially be classified as a **secondary diagnostic** component during the early operational periods shifting to a **primary** monitoring component in the later operational and decommissioning phases.

Upon decommissioning of the TMF, piezometers should be installed in any constructed embankments with additional peizometers located near the TMF to monitor seepage and shallow groundwater movement.

The geophysical methods detailed in the TSN 8.0 are also applicable for determining the general extent of groundwater seepage plumes from TMFs. At a minimum they should be useful in determining borehole locations. Additional information on groundwater monitoring is provided in Chapter IV, with information on specific sampling techniques provided in the TSN 7.2, 7.3, 14.3 and in MEND 4.5.1 (1994).

<u>*Pore Water*</u>

A comparison of tailings pore water quality to TMF seepage water or local groundwater quality can be useful in understanding subsurface flowpaths from a TMF and the dynamics of AMD generation as it migrates downward and laterally through the tailings.

Pore water monitoring within the tailings should be classified as a **secondary diagnostic** component:

- if predictive modelling suggests a high AMD potential; and/or
- if significant acid generation has been detected.

Tailings porewater monitoring in the unsaturated or vadose zone may be completed using core extraction procedures (TSN 14.2.3) or suction lysimeters (14.2.4). Porewater monitoring in saturated tailings can be completed using peizometers (TSN 14.3). For additional information on porewater monitoring of surface tailings in flooded or subaqueous TMFs see ChIV:3.6.1.

Consideration should also be given to installing peizometers (TSN 14.3) to determine the pore water flow rate. This information is required to determine sampling frequency and for predictive AMD modelling purposes.

*Tailings Characteristics*

Internal tailings compaction and pore pressure monitoring should be completed during the operational phase using geotechnical instruments such as piezo-cone penetration testing (TSN 8.5). This monitoring should continue into the decommissioning phase until suitable tailings compaction[8] has been achieved. This will determine water infiltration rates and help establish long-term AMD generation potential. TMFs constructed with side and bottom drains should compact more rapidly. These measurements should be completed monthly (BC AMD 1989).

### 2.3.4 Investigative Monitoring

AMD investigative monitoring for TMFs tends to focus on verifying and fine-tuning AMD modelling predictions to increase confidence in the long term predictive modelling for decommissioning. Investigative monitoring intensity will increase just prior to and/or immediately after closure (decommissioning baseline) to assess the future potential for AMD production and to identify and establish any long-term control facilities.

- Tailings solids should be collected every 3 to 5 years for predictive AMD tests (TSN 2.0, 6.0) to confirm or re-assess previous predictive tests and modelling (TSN 3.0).
- Tailings pore water (TSN 14.2, 14.3) data may be collected to provide indications of:
  - the level of acid production when oxidizing reactions are occurring; and
  - acidity and reaction products in the pore water inventory that are potentially available to migrate.

---

[8] Based on relative comparisons to the hydraulic conductivity of the surrounding strata.

- Pore gas monitoring (TSN 3.1, 3.2)may be required to support research and modelling of the processes taking place within the tailings, and the effects of changed conditions (e.g. construction of a dry cover (TSN 3.1-3.5) or subaqueous disposal options).
- Geophysical monitoring techniques (TSN 8.0) may be employed for groundwater assessment or plume detection.

## 2.4 Mine Workings

### 2.4.1 Background

Mining methods fall into two broad categories: surface pit mining and underground mining. Open pit mining involves the removal of the overburden and the barren overlying rock to expose the orebody. This excavation is generally completed in a series of step-like layers (benches), which are expanded outwards to the designated perimeter of the pit (Figure III:2.4-1a).

In underground mining a minimum amount of overburden and barren rock is removed to provide a point of access to the ore. Horizontal openings (e.g., adits and tunnels) and vertical openings (e.g., shafts and raises) are excavated to provided access for workers and equipment, routes for extracting the ore and ventilation networks. The selected underground mining method depends on the geometry of the orebody and the structural integrity of the ore and the surrounding rock. The voids created by the removal of the ore may be backfilled with broken rock or tailings. Fractures resulting from subsidence after decommissioning may extend to the surface providing an additional linking pathway to the environment (Figure III:2.4-1b).

Both open pits and underground mines commonly incorporate some form of local dewatering program (e.g., dewatering wells, networks of sump pumps, gravity drains, etc.) to minimize the inflow of water (primarily groundwater) to the mine workings. Upon decommissioning the dewatering program ceases and the mine workings (open pit or underground network) are permitted to flood.

### 2.4.2 Pathways

There are three main potential sources for AMD associated with mine workings (Figures III:2.4-2 and III:2.4.3). These are:

- zones of sulphide mineralization outside excavations which are exposed to oxidation when the water table is lowered;
- sulphide mineralization in the exposed walls of the pit or underground excavations (i.e., including additional surface area exposure due to erosion, bench failure, etc); and
- additional sulphide-bearing materials deposited in a pit or underground mine such as waste rock, tailings materials, or backfill material.

Oxidized sulphide deposits exposed by dewatering are generally the lowest contributor to AMD since they exist in intact formations with limited infiltration of water and oxygen. AMD generated from this source is carried by subsurface flow to the groundwater table. During the operational phase of the mine this groundwater tends to flow towards the dewatering wells (if present) or to the open pit or underground mine. The water from the dewatering wells is usually pumped for treatment, and the water entering the mine works becomes a part of the regular mine water management operations.

The primary sources of AMD from mine workings arise from air and water entering the mine (e.g., infiltrating precipitation, groundwater, drilling water) and interacting with sulphide minerals associated with the exposed faces of the mine (pit or underground) and any fractured rock material stored in the mine works (e.g., waste rock, backfill). During the operational phase potentially contaminated waters flow downward (surface flow or shallow sub-surface flow) to collection points (bottom of the pit or underground collections points) and are pumped to the surface. Alternatively, if the topography allows, the water may be directed to the surface through gravity drains and collected. In most cases, mine water brought to the surface is pumped to a treatment facility or used in the milling process.

Some AMD generated in the mine works may enter the local groundwater system. However, dewatering activities surrounding and within the mine tend to draw groundwater flow back into the mine.

At closure, mines are typically allowed to flood, usually through natural recharge. Water levels in flooded underground mines or open pits often rise to pre-development watertable levels (below the surface, shaft/raise collar, and pit crest elevations). In some instances, mine water may discharge through mine openings at the surface (i.e., adits, shafts, raises, glory holes, crown pillar stopes) and result in another potential AMD pathway to the environment. Initially, flooding results in a flushing of stored oxidation products increasing AMD contamination in the flooded mine waters. Additional

AMD generation should decrease as the submergence of most or all exposed sulphide minerals inhibits their oxidation.

With the cessation of the dewatering program, the groundwater flow regime tends to revert back to pre-operational flow patterns. Thus, while the groundwater flow during the operational stage tends to discharge into the mine, upon closure the alteration of the groundwater flux through the mine may result in AMD waters entering the regional groundwater system.

### 2.4.3 Compliance and Diagnostic Monitoring

The monitoring categories applicable to mine workings are summarized in Table III:2.4-1. **Compliance** and **diagnostic** monitoring for mineworks during the **operational phase** focus on the quality of the mine water collected, and pumped or drained to the surface. Depending on the local hydrogeology, groundwater monitoring may also be necessary during the operational phase. During the **decommissioning phase**, monitoring of the flooded mine works shifts to monitoring the quality of the *in situ* flooded mine water and any surface releases. Additional emphasis is placed on groundwater monitoring as outward groundflow becomes a factor. Compliance stations will likely be required at all points of surface water release to the environment. If this is not the case the operator should locate a primary diagnostic station at these points.

#### 2.4.3.1 Identifying Sampling Locations and Periods / Frequency

Schematics of the AMD pathways for a generic open pit and an underground mine are provided in Figures III:2.4-2 and III:2.4-3, respectively, along with recommended priorities for monitoring these pathways. The classification system is defined in ChIII:2.1 with the locations subdivided into Compliance, Primary, and Secondary Stations.

***Primary and Secondary Stations***

Mine water monitoring is part of the minesite water quality monitoring program and is essential for proper water quality management. Mine water management plans involve the control of water within the mine including distribution, collection, dewatering, and protection against inrush.

*Surface Waters*

**Primary** surface water quality stations should be located at all points of release to the environment upflow of the compliance stations during the operational and decommissioning phases. Any points of release to the environment not designated as compliance stations should be classified as primary diagnostic stations. This would involve:

- monitoring water directly released to the environment from the mineworks
  - Operational Phase:
    drainage from gravity drains or from mine openings (e.g. adits);
  - Decommissioned and Flooded:
    spillway discharge from pits or surface discharge from underground works (e.g., adits, shafts, glory holes); and
- end of pipe monitoring of water released from associated AMD treatment facilities.

To optimize treatment protocols, a **primary** or **secondary diagnostic** monitoring station, depending on the sensitivity of the treatment process, is required to analyze water pumped to any treatment system. This may involve water pumped:

- from the bottom sump(s) of open pits;
- from various underground sumps (e.g., mine voids);
- from any points of surface discharge (e.g., gravity drains, flooded pit spillways, surface discharge from flooded mine openings);
- from dewatering wells; and
- from flooded works.

The minimum recommended sampling frequency for sump collections is monthly (BC AMD 1989). However, it is recommended that a seasonally stratified sampling routine (see ChIII:2.5) be adopted for open pit sump collections due to seasonal flushing of AMD from open pits. A fixed sampling frequency is suitable for monitoring sump water from underground works as the primary water source is groundwater which is less influenced by seasonal discharge patterns. Further information on water quality sampling frequencies is presented in ChIII:2.5.

In flooded (decommissioned) open pits, *in situ* water quality should be assessed using a **minimum** of one deep water station with sample points distributed vertically through the water column. The number and the depth distribution for these sampling horizons will depend on the total depth of the pit and whether a thermocline or chemocline has developed.

If the pit waters stratify, samples should be collected from the surface (0.5 m), immediately above the epilimnion, immediately below the epilimnion, mid-hypolimnion and 0.5 to 1 m above the sediment surface. (See ChIV:2.0 for further information). If no thermocline develops, then samples should be collected from the top, middle, and bottom of the water column. Similarly, if a chemocline develops sampling should be distributed to assess the chemical variability within the water column.

The vertically distributed samples should **not** be composited. Initially, sampling frequency should be monthly and can be extended (e.g., seasonal/quarterly) once the hydrological patterns of the system are understood. Consideration should be given to stratifying sampling frequency if strong seasonal patterns are evident in precipitation or other inputs to the flooded pit.

*<u>Groundwater</u>*

**Primary** groundwater monitoring stations are required if aquifers intersect with mineworks. A minimum sampling program would involve one up-gradient and one down-gradient well for each hydraulically related stratum associated with the mining facilities (BC AMD 1989). Higher hydraulic conductivity of the subsurface stratum and increased AMD potential, will require a denser groundwater network and a greater sampling frequency.

During the operational phase:

- a **primary** groundwater quality and flow monitoring program should be associated with the dewatering wells. This water may require treatment and should be monitored to identify the specific treatment requirements; and
- the groundwater flow regime within the perimeter of the dewatering system should be monitored (**secondary diagnostic**) using piezometers to confirm that groundwater flow from the pit is minimal and that water quality monitoring is not yet required.

After flooding and during the decommissioning phase the groundwater monitoring network becomes a high priority as groundwater flow from the mineworks may commence with an initial AMD pulse until previously exposed sulphides are flooded.

- Groundwater flow monitoring should be maintained and upgraded to a **primary** monitoring component incorporating water quality analysis.
  - The original network of dewatering wells serves to supplement the groundwater monitoring program.

BC AMD (1989) recommends groundwater monitoring (other than water pumped to a treatment facility) be completed every six months. Consideration should be given to modifying this sampling frequency to reflect the hydraulic conductivity of the strata being monitored. Additional information on groundwater monitoring is provided in ChIV:1.0 and 2.0, with information on specific sampling techniques provided in the TSN 14.3 and in MEND 4.5.1 (1994).

### 2.4.4 Investigative Stations

As in waste rock and TMF monitoring, investigative monitoring should focus on verifying and fine-tuning AMD model predictions to increase confidence in the long-term predictive decommissioning modelling. Such programs tend to be more site-specific such that, what is considered investigative at one facility may be a primary monitoring component at another facility due to differences in AMD risks. Investigative monitoring intensity should increase just prior to and/or immediately after closure (decommissioning baseline) to assess the future potential for AMD production and to identify any long-term control facilities.

Additional investigative monitoring may involve:

- periodic re-assessment of AMD potential for specific mine walls
  - using AMD prediction protocols outlined in ChIII:4.0. (e.g. Miller et al. 1991a)
- collection of seepage waters from specific mine walls or sections to confirm predictive AMD assessments and to direct non-acid and acid flows to different sumps to minimize treated water volumes; and
- the use of geophysical monitoring techniques (TSN 8.0) for groundwater assessment or plume detection.

## 2.5 Design Considerations For Source Surface Water Programs

Surface water quality is the core component of any source monitoring program. Station locations tend to vary depending on the source and have been previously discussed in this section. Source water quality monitoring programs may be subdivided into the monitoring of:

- water with the **potential** for AMD contamination; and
- waters **contaminated** with AMD.

### 2.5.1 Potential AMD Surface Waters

Monitoring untreated drainage or seeps serves as an early warning system to determine whether the buffering plateau (Figure II:1.2-1) of a potential source has been exceeded leading to the commencement of, or an increase in, the release of AMD contaminants. Waste management decisions can then be identified which will range from increased diagnostic or investigative monitoring to the initiation of various remedial or control options (BC AMD 1989).

Decisions relating to the interception and treatment of drainage waters should be based on peak concentrations. Due to the hysteresis effect (ChII:1.4), a monitoring program consisting of single monthly samples has a high probability of missing AMD flushing events and is therefore not considered adequate. Since the main loads of AMD contaminants are flushed during moderate to high discharges following a low discharge period, sampling should be concentrated within these flushing period(s). Thus, the only reliable way to determine peak concentrations is to collect a minimum of daily samples during periods of high AMD release.

Probable peak flushing periods should be identified from seasonal discharge data for streams on or adjacent to the site (ChIV:1.1) and precipitation and snow melt records. This will serve to roughly delineate the most likely periods of peak AMD release (Figure II:1.4-2). The importance of capturing the first flushing event means the monitoring commencement date must be flexible and determined by the on-site operator. For example, as the roughly delineated sampling period approaches, the on-site technician should be prepared to commence the daily sampling in response to the first significant flushing event (rock pile snow melt or significant rainfall).

Daily sampling should continue until a peak concentration has been detected followed by evidence of a number of days of declining concentrations. Hence, relatively rapid analytical results are required if the data are to be used to identify when sampling may cease. To achieve this, the chemical analysis may be completed using the mine's laboratory facility. Since peak concentrations are the measurement of interest, precision near the detection limit for metals is of secondary concern relative to the need for rapid feedback for decision making. If analytical error associated with the mine laboratory is a concern, it may be quantified with a preliminary study based on split sample comparisons with another laboratory. Since preserved samples for metal analyses may be held for up to six months, replicates (≈3 to 5) should be collected each sampling period. One sample would be analyzed and the other replicates held until the peak single sample concentration was identified. The replicates from the peak sampling day and the adjacent pre- and post-peak days may then be analyzed to confirm the peak concentration and provide a measure of the instantaneous variation.

#### 2.5.1.1 Parameters of Interest

The most efficient approach to monitoring source water quality would be to employ a tiering strategy for analyte selection. Parameter selection will tend to be site-specific; however, some generalities apply and are discussed here. When AMD generation is of **low concern**, analyses could be limited to Tier I parameters such as **pH**, **conductivity**, **acidity, alkalinity**, and **sulphate**. Trace metals would be considered Tier II parameters and be measured when decreases in pH and/or alkalinity or increases in conductivity and/or sulphates are detected. The trace metals of interest would be those previously identified during the AMD predictive assessment program. Discharge should be measured for each sampling event (see ChIV:1.1, and TSN 7.1).

When predictive AMD assessment has identified a source as of **moderate to high AMD potential**, trace metal(s) should be added to the previously identified Tier I parameters. If site-specific predictive modelling has identified certain trace metals as early precursors[9] of AMD generation (e.g., Al and Zn) they should be incorporated as Tier I parameters with additional metals classified as Tier II. Should there be evidence of increased concentrations of the selected Tier I trace metal(s), then a complete analysis of all of the trace metals identified as potential leachates for the site should be completed (Tier II trace metals, possibly an ICAP scan). This should also trigger the initiation of a detailed AMD preliminary investigation for what should now be considered an active AMD source. This investigation will serve to provide the information required to develop the long-term monitoring program for the now identified AMD source. Details for this are provided in the following section.

### 2.5.2 Identified AMD Surface Waters

When AMD source(s) of moderate to high concern have been identified by the monitoring program, the minesite surface water management plan should be adapted to separate contaminated drainage from uncontaminated surface flows, as proposed in the contingency plans of the Final Mine Plan. Separation of contaminated and uncontaminated discharges will isolate contaminated water into a single AMD pathway and thereby minimizes the volume of water requiring detailed monitoring or treatment. A primary diagnostic station should be located at a point downstream of the final inflow point to this AMD pathway to facilitate detailed discharge monitoring. Once AMD has been

---

[9] Zn entering solution due to acid generation tends to remain in solution after neutralization.
Neutralization with $Al(OH)_3$ (gibbsite) mobilizes aluminium, e.g., $Al(OH_3) + 3H^{+} \rightarrow Al^{+3} + 3H_2O$
Hence, both these metals serve as early indicators of neutralized AMD prior to evidence of a significant decrease in pH in the drainage.

identified and localized, a preliminary intensive monitoring program will be required to confirm the seasonal or discharge related dynamics of the AMD and determine loadings and peak concentrations. This information will support decisions regarding management and/or treatment of the discharge and will aid in the design of the future monitoring program.

### 2.5.2.1 Preliminary Investigations

Without an intensive preliminary survey, the developed monitoring program may be costly, inefficient and inadequate to meet the data requirements necessary for informed management decisions. A poorly designed monitoring program may result in failure to detect lethal pulses of contaminants prior to release to the receiving environment, or in the design of inadequate treatment facilities. The monitoring program should be developed in consultation with a biostatistician or other personnel familiar with the statistical design of water monitoring systems.

The basic principles of developing a preliminary monitoring program for AMD contaminated waters are provided in this section. Information on the statistical terms used below are provided in ChII:2.5. For more detailed discussions on statistical designs for monitoring water quality of acid mine drainages, see BC AMD (1990b). The preliminary investigation for the AMD contaminated waters should be designed to identify the following information for the primary monitoring stations.

***Parameters of Interest***

The preliminary investigation should include all of the Tier I and Tier II parameters discussed in ChIII:2.5.1.1.

***Stratification***

The preliminary investigation identifies factors that may influence the mean and variance of the AMD water quality variables. The rate of release of contaminants from AMD sources is usually seasonal and discharge related; therefore the preliminary sampling program should be subdivided into strata representing this dependence relationship. An example would be a sampling program initially subdivided into four strata consisting of:

- snow melt strata;
- summer strata;
- autumn strata; and

- winter strata, as shown in Figure II:1.4-2.

The discharge data collected at potential AMD sources prior to detection of AMD, and the hydrology/meteorological data collected as part of the general mine monitoring program (ChIV:1.1) will aid in the initial stratification.

Within each stratum, a number of samples should be collected in order to calculate the mean and variance for the AMD components (see ChII:2.5) and all suspected co-factors such as discharge and temperature. Sampling frequency must be of sufficient intensity to capture the range of variability within each of the stratum. This will generally require a **minimum** of weekly water quality samples for one year, assuming periods of no flow (dry or frozen) do not occur.

The ideal scenario involves the use of an *in situ* continuous probe attached to a data logger, with continuous discharge data also collected using the options discussed in ChIV:1.1 and TSN 14.1. Continuous data logging probes which are capable of measuring pH and conductivity can be purchased commercially. A list of parameters which can be determined using continuous time probes and data loggers is presented in Table III:2.5-1. Additional AMD parameters of interest (see ChIII:2.5.1.1) should be sampled using the previously mentioned minimum schedule.

***Instantaneous Variation (Replication)***

To determine how representative a single sample or mean estimate is for that specific sampling time, the within station instantaneous variation must be determined by collecting sample replicates (3 to 5 replicates). Identifying within-station instantaneous variation serves two functions:

- it allows for the exact determination of the number of replicates required for the long-term monitoring program (power analysis); and
- allows the separation of instantaneous variation from temporal variation which would otherwise lead to a monitoring design requiring more frequent sampling and, hence, more long-term expense (BC AMD 1990b).

If AMD components within the discharge are homogeneously mixed, then the variation between the replicates will be low. In such scenarios, a single sample is a suitable representative of the water quality at that moment in time. In most situations instantaneous variation need only be determined during one sampling period for each stratum.

***Autocorrelation***

To identify the optimal sampling frequency the autocorrelation in the system must be determined. The objective is to identify the time interval (*lag effect*) between sampling events that is required for sample concentration to be uninfluenced by the concentration of the previously collected sample. The *time lag* required for a specific parameter can only be identified by sampling more frequently than the duration of the time lag. This should not be a problem for pH or conductivity if a continuous data logger is used. If sampling of the additional AMD parameters was completed weekly it would not be possible to identify time lags of less than one week; therefore more intensive sampling should be considered for the preliminary investigation.

### 2.5.2.2 AMD Monitoring Program

Based on the results from a comprehensive preliminary investigation the monitoring program can be designed to meet site-specific requirements. These may involve accurately quantifying the volume and composition of the AMD to minimize treatment requirements, optimize the treatment process, and to meet compliance criteria. Another common requirement in water quality monitoring involves estimates of total contaminant loadings to the environment. The former examples require study designs focussed on close monitoring of peak concentrations, while the latter may be suitable and more economically achieved with a design directed towards accurate determination of mean concentrations. Examples of these different design requirements are provided in the following sub-sections.

### *Peak Concentrations*

Accurate monitoring peak concentrations may be of interest when monitoring the inflow to a treatment facility, in order to refine the treatment process to achieve the most cost efficient (e.g., minimizing reagent expenses and transportation costs) and environmentally sound results. Peak concentrations are also of interest to monitor for **alert levels**, which act as early warnings of values approaching permit concentrations for downstream compliance stations. These alert levels should be set at concentrations which provide enough response time for operational adjustments or the initiation of intervention plans (e.g., redirect to a temporary holding pond).

Accuracy in monitoring peak concentrations requires intensive monitoring. A cost effective approach may involve a continuous proxy monitoring design. The results of the preliminary monitoring program should be analyzed to identify a parameter which may be monitored on a continuous basis that also exhibits a strong correlation with the parameter of concern (e.g., a certain trace metal) (Morin and Hutt 1993). The most likely candidate is pH. The correlation is usually insufficient to confidently calculate trace metal concentration directly from the pH value. However, pH is usually a reliable enough indicator to serve as the *trigger* to initiate sampling of water for trace metal analyses. For example, preliminary survey results may show pH values of less than 4.5 are correlated with copper concentrations approaching levels of concern. In such a scenario, sample collection would be initiated for copper analysis when the pH declined to values of 4.5 or less.

Collection of samples for trace metal analyses could involve one of two procedures. The system could be designed to signal an operator (via a visual or audio signal) to manually collect a grab sample, or monitoring equipment may be purchased that collects and stores the samples for removal at a later time (note preservation and holding times in Table IV:2.1-2 and TSN 13.2). Chemical analyses may be restricted to samples collected during the peak pH events, thereby saving a great deal in expensive analytical costs. The expense of self-sampling monitoring equipment must be balanced against the manpower requirements of the former monitoring design. Additional cost savings will be realized due to the benefits inherent in an efficient treatment system tailored to respond to the inflow dynamics of the AMD.

### *Accurate Mean Concentrations*

The most accurate method of determining contaminant loads to the environment would involve continuous time measurements and data logging of the contaminants. However, as previously discussed, this is not possible for trace metals. Determination of accurate mean concentrations is usually the most realistic and cost efficient method for estimating monthly or annual contaminant loadings to the receiving environment. In systems which exhibit extreme fluctuations in contaminant concentrations, some incorporation of peak concentrations may also apply. An example of developing monitoring programs for total AMD loadings is provided in BC AMD (1990b) with the basic process subsequently summarized. Such investigations would be located at the points of release of treated or collected AMD to the receiving environments and would therefore usually coincide with compliance stations.

A preliminary investigation should be completed as described in ChIII:2.5.2.1 to identify the variance, frequency distribution, and autocorrelation in the water quality. Based on these data suitable strata should be identified (e.g., seasonal, discharge, pH, etc.) and a sampling program developed with sampling intensity distributed amongst the strata in a manner which provides approximately equal variance. The intensity of the sampling program may then be set to achieve a designated level of confidence for an estimation of the means. The minimum time interval between sample collection required to collect independent samples is determined by identifying the level of autocorrelation in the preliminary database. The sampling dates should then be randomly allotted within the sampling period, rejecting dates conflicting with the minimal time interval (BC AMD 1990b).

If the system exhibits a great deal of fluctuation, accurate load estimates may also require incorporation of peak contaminant concentrations along with the mean estimates. This would require additional data collection as described above for determining peak concentrations. Decisions on incorporating peak concentrations will have to be made upon determination of the additional cost in relation to the increased confidence in the estimates of contaminant loadings.

Accurate discharge measurements are also required to determine total loadings. The ideal situation would involve the use of a permanent discharge data logging system. See ChIV:1.1 for more information on appropriate procedures for determining discharge.

With these data contaminant loadings may be calculated as a function of concentration and the volume discharged. When total loadings are determined using concentrations and discharge values with accompanying error estimate, the error estimate of the contaminant loading may also be determined. This allows for additional confidence in modelling estimates.

### 2.6 Structural Integrity

Periodic geotechnical monitoring should be completed to determine the structural integrity of the various AMD sources at a minesite. Monitoring of the structural integrity of rock piles should involve such factors as consolidation/settlement and slope stability to assess the potential for hydrological changes within the structure and to prevent slumping and toe collapse (BC AMD 1989). Attention should be placed on determining internal particle size of large waste rock dumps, as recent studies suggest that internal weathering and chemical dynamics can result in rapid transition from large to small particles sizes. This has significant implications for shear stress and structural stability (Herasymuik et. al. 1996). Therefore, a pre-decommissioning assessment of the long-term stability of the rock pile should be completed.

The structural integrity of the TMF and any mine workings is also routinely investigated during the operational phase. A detailed pre-decommissioning assessment of the long-term stability of a TMF, worked out open pit, underground mine, and any other associated control structures should be completed in order to quantify the relative risk of structural failure.

### 2.7 Section Summary

#### Generic Surface Water Monitoring

**Primary diagnostic** monitoring stations should be located at all points of surface water release upstream of compliance stations and any non-compliance points of release to a natural water course. Primary diagnostic stations are also required to monitor contaminated water flowing into, and treated waters flowing out of AMD treatment facilities.

**Secondary diagnostic** surface water stations are required when AMD concerns are high. They are established to delineate drainages from different potential AMD sources, and to monitor between primary (source) and compliance stations when there is the potential for significant changes in water quality between the two stations.

#### Generic Groundwater Monitoring

**Primary groundwater** monitoring stations are required if groundwater interacts with the potential AMD source. Where groundwater is a concern, samples should be collected from daylights and/or well(s) located along the flowpath with a minimum of one up-gradient and one down-gradient well for each hydraulically related stratum potentially impacted.

### Additional Generic Monitoring Components

Additional secondary monitoring requirements vary depending on the AMD source, i.e., waste rock, TMF or mineworks. Investigative monitoring should focus on verifying and fine tuning AMD model predictions to increase confidence in the long-term predictive decommissioning modelling.

### Waste Rock

Primary **surface water** quality stations specific to rock piles should be located:

- at the toe of waste rock piles or in the surrounding collection ditch; and
- at any points of discharge from the drainage ditch to a natural water course (e.g., stream on or off property). This should be accompanied by an upstream reference station if feasible.

Primary or secondary **groundwater** monitoring for rockpiles is only required if the shallow groundwater system interacts with the rock pile.

**Internal monitoring** of rock piles is generally required for investigative purposes either in response to active acid generation or for on-going fine-tuning of predictive AMD modelling.

### Tailings Management Facilities

Primary **surface water** quality stations should be located at all points of release to the environment up-flow of the compliance stations. This involves:

- monitoring of embankment seepage waters;
- monitoring water directly released to the environment from the TMF.
  (assuming this is not already a compliance station)

The chemical composition of the **slurry** (both tailings solids and liquid effluent) should be monitored prior to deposition.

Water pumped from the TMF should be monitored. This includes:

- water pumped from surface ponds or flooded TMFs; and
- water pumped from TMFs with side and bottom drain systems.

**Groundwater** monitoring stations are required if aquifers intersect with the TMF. During the early operational phase groundwater may be a secondary diagnostic component becoming a primary component in the later operational and decommissioning phases.

Piezometers should be installed in any constructed embankments with additional piezometers located near the TMF to monitor seepage and shallow groundwater movement. Geophysical methods may be used to support seepage monitoring.

**Pore water** monitoring (quality and flow) within the tailings should be classified as a secondary diagnostic component:

- if predictive modelling suggests a high AMD potential; and/or
- if significant acid generation has been detected.

Internal **tailings compaction** and **pore pressure** monitoring should be completed during the operational phase and continued into decommissioning until suitable tailings compaction has been achieved.

TMF **investigative monitoring** will focus on verifying and fine-tuning AMD modelling predictions to increase confidence in the long term predictive modelling for decommissioning. Investigative monitoring intensity will increase just prior to and/or immediately after closure (decommissioning baseline) to assess the future potential for AMD production and to identify and establish any long-term control facilities.

**Mineworks**

Primary **surface water** quality stations should be located at all points of release to the environment up-flow of the compliance stations. This would involve:

- monitoring water directly released to the environment from the mineworks
  Operational Phase:
  - drainage from gravity drains or from mine openings (e.g. adits);
  Decommissioned and Flooded:
  - spillway discharge from pits or surface discharge from underground works.

Water pumped from the TMF should be monitored. This includes:

- from the bottom sump(s) of open pits;
- from various underground sumps (e.g., mine voids);
- from any points of surface discharge (e.g., gravity drains, flooded pit spillways, surface discharge from flooded mine openings);
- from dewatering wells; and
- from flooded works.

*In situ* water quality of flooded mineworks should also be monitored.

Primary **groundwater** monitoring stations are required if aquifers intersect with mineworks. During the operational phase:

- a groundwater quality and flow monitoring program should be associated with the dewatering wells; and
- the groundwater flow regime within the perimeter of the dewatering system should be monitored (secondary diagnostic) using piezometers.

After flooding and during the decommissioning phase, groundwater flow monitoring should be maintained and upgraded to a primary monitoring component incorporating water quality analysis. Geophysical methods may be used to support this program.

Short-term **investigative monitoring** should involve:

- periodic re-assessment of AMD potential for specific mine walls

- collection of seepage waters from specific mine walls or sections to confirm predictive AMD assessments and to direct non-acid and acid flows to different sumps to minimize treated water volumes.

Additional investigative monitoring should focus on verifying and fine-tuning AMD model predictions to increase confidence in the long-term predictive decommissioning modelling.

### Design Considerations For Source Surface Water Programs

Monitoring untreated drainage or seeps serves as an early warning system identifying whether the buffering plateau of a potential source has been exceeded leading to the commencement of, or an increase in, the release of AMD contaminants.

#### *Potential for AMD Generation*

Decisions relating to the interception and treatment of drainage waters should be based on peak concentrations. This requires a minimum of daily sample collection during periods of high AMD release. When AMD generation is of **low concern**, analyses could be limited to Tier I parameters. Trace metals would be Tier II parameters measured when decreases in pH and/or alkalinity or increases in specific conductivity and/or sulphates are detected. When predictive AMD assessment has identified a source as of **moderate to high AMD potential**, trace metal(s) should be Tier I parameters. Tier II confirmation of a shift to a higher AMD status should trigger the initiation of a detailed AMD preliminary investigation for what should now be considered an active AMD source.

#### *Confirmed AMD Generation*

Confirmation of acid generation should lead to:

- containment and localization of the AMD;
- initiation of a preliminary intensive monitoring program to,
  - confirm seasonal or discharge related dynamics
  - determine peak concentrations and estimate loadings; and
- the design of a long-term monitoring program for the identified source.

Monitoring programs addressing monthly or annual contaminant loadings require the determination of accurate mean contaminant concentrations using a stratified sampling design.

Monitoring programs concerned with potential effects of peak AMD releases to sensitive habitats or organisms require more intensive sampling with continuous monitoring during periods of potential high release.

## 2.7 Section Bibliography

**Waste Rock and AMD**
Bennett. and Pantelis 1991
Harries and Ritchie 1981, 1983, 1985
Kwong 1991
MEND 1.11.1 1991
MEND 1.14.2 1997
MEND 1.14.2c,d,e,f,g 1994
MEND 1.19.1 1994
MEND 1.22.1a 1994
MEND 2.31.1a 1992
MEND 1997
Morin and Hutt 1995
Robertson and Barton-Bridges 1990
SEPS 1992
SRK 1993
Stromberg et al. 1995

**Subaqueous Deposition**
Brassard et al. 1996
MEND 2.11.1a 1989
MEND 2.11.1a 1990a,b,c,d

**Mineworks**
Miller et al. 1990
Arnesen et al. 1991
Miller et al. 1991a
Bowell et al. 1995

### Waste Rock Sampling

MEND 4.5.1-1 1994
MEND 4.5.1-2 1994

### Tailings and AMD

MEND 1.17.1a 1986
MEND 1.17.1b 1987
MEND 1.17.1c 1988, 1990
MEND 1.17.1d 1990
MEND 1.21.1a 1990
Fortin et al. 1995

### Tailings and Sampling

Hornsby et al. 1991
MEND 4.1.1 1989
NUTP 1985

### Source Vadose Zone Monitoring

Bond and Rouse 1985
Bumb et al. 1988
Robbins and Gemmell 1985
Bond 1995
Wilson et al. 1995

### Source Hydrogeology

Hawkins and Aljoe 1991
Aljoe and hawkins 1991
Yanful and St. Arneaud 1991
Coggans et al. 1991
Lambeth et al. 1991

### Geophysical Source Monitoring

King and Sartorelli 1991
MEND 4.6.1 1994
MEND 4.6.3 1994
Yanful and St. Arneaud 1991

# CHAPTER IV - RECEIVING ENVIRONMENT MONITORING

The objectives of receiving environment monitoring are two-fold:

1) monitoring contaminant inputs to the environment, and
2) identifying any changes in the biota which can be attributed to the contaminant input.

## *Monitoring the Abiotic Receiving Environment*

Objective 1 requires monitoring of the contaminant concentrations in the abiotic components of the environment which serve as the connecting pathway to biota. Tier I abiotic monitoring components should consist of surface water quality and sediment chemical analyses using Tier I parameters (see Hydrology: ChIV:1.0, Water Quality: ChIV:2.0, Sediment: ChIV:3.0).

The selection of biological components for objective 2 is somewhat more complex. This requires consideration of the site-specific contaminants, the range of biota exposed, and the possible toxicological implications of exposure to specific biota. The following paragraphs should help to clarify objective 2 considerations.

## *Monitoring the Biotic Receiving Environment*

It is generally accepted that the effects of chronic stress are manifested at lower levels of biological organization before there are expressions at the population, community or ecosystem levels. Effects are expressed first at the molecular and biochemical levels as changes in the functioning of enzymes, cell membranes or genetic material. Should the stressor persist, these changes may produce a series of structural and functional responses that may impair integrated processes such as hormonal regulation, metabolism, osmoregulation, and immunological functions. If significant, these effects may alter an organisms rates of growth, reproduction and survival, ultimately resulting in irreversible, negative effects at the population, community and ecosystem levels (Adams 1990). A bewildering range of biological monitoring techniques, operating at various stages along this chain of responses, has been proposed in the literature.

One way of organizing these options is shown in Figure IV:0-1. In this figure, the response chain is spatially distributed along a **horizontal** continuum representing the time interval between exposure and a measurable effect at a specific organizational level. The organizational levels of the response chain are **vertically** ordinated along a continuum representing the significance or likelihood of an effect having a long-term detrimental impact on the ecosystem as a whole. The spatial pattern that arises is one where rapidly responding (seconds to minutes after exposure) sensitive indicators (low levels of organization) have low ecological relevance, while slowly responding (biological generations or years), hence poor early warning indicators, have high ecological significance.

This results in an organizational continuum that is bounded by community and population indices at the top and biochemical indicators at the bottom (Figure IV:0-1). The debate concerning bottom-up or top-down monitoring approaches is based on this spatial distribution of developed and developing biological indices.

*Top-down vs. Bottom-up*

The **bottom-up strategy** monitors for evidence of impacts at the lower organizational levels. Positive results trigger additional investigations to determine whether the measured effects are translating upwards and resulting in effects at the population/community level. The bottom-up strategy uses Tier I monitoring components/parameters (e.g., biochemical measures) that are highly sensitive to perturbations (anthropogenic and natural). As a result they allow for the initiation of remedial action(s) to prevent translation of impacts to higher levels of organization (Figure IV:0-1). However, the sensitivity of this approach is one of its major drawbacks. With the present level of knowledge, it is difficult to separate natural variation from actual mining related impacts. In addition, many of the detected impacts will pose no threat to ecosystem integrity, as they may never be reflected at the higher levels of organization (e.g., population or community) due to the compensatory capabilities of individuals and populations. In operational terms, this approach will increase the number of unnecessary additional monitoring investigations, mitigative projects, and shut-downs.

The **top-down strategy** employs community monitoring techniques as Tier I monitoring components. Evidence of community/population effects results in the initiation of Tier II and III components involving lower levels of organization to confirm the effect, and identify the causal agent(s) (Figure IV: 0-1). This restricts the detection of impacts to those likely to be of high ecological relevance and has the greatest potential to affect ecosystem health significantly. However, top-down monitoring detects impacts only **after** they have operated at the upper levels of organization. Thus, this approach provides limited response time, thereby, increasing the chances of identifying impacts only after significant damage has been done to the ecosystem. In operational terms, this increases the risk of serious long-term shut-downs, significant regulatory fines, and the institution of restrictive (when the actual causal agent has not yet been identified) environmental regulations to prevent a re-occurrence.

To balance the pros and cons of the two monitoring strategies, it is recommended that AMD monitoring incorporate both top-down and bottom-up components, using organisms which vary in their perceived acceptable levels of risk. For example, an after-the-fact detection of an effect (e.g., species loss) to the fish community would be considered unacceptable; however, this is not necessarily the case for the benthic macroinvertebrate community. The loss of an invertebrate species or assemblage may not result in a significant change in system functioning (BC AMD 1990c), and hence may have little influence on the higher trophic levels. The detection of the impact at the benthic macroinvertebrate community level provides response time for further investigations (Tier II and III) to determine the possible significance of the detected effect on fish populations. Thus, it is recommended that the benthic macroinvertebrate community be used as a Tier I community monitoring component (ChIV:4.2).

Tier I monitoring should also include a fish sentinel species, using monitoring tools operating at the level of the whole organism or lower (ChIV:5.0). This will improve response time for the protection of fish species, and addresses concerns over the difficulties in justifying significant changes in mining operational procedures if evidence of an effect is restricted to algal or invertebrate communities (Moore and Ramamoorthy 1984). In recognition of the failure of fish (vertebrates) to meet a number of the criteria preferred in a sentinel organism, consideration should be given to adopting an additional organism such as bivalves or crayfish (when available) for this purpose. Unfortunately, while a great deal of work has been directed towards the use of invertebrates (bivalves and crustaceans) in marine systems, the use of freshwater invertebrate sentinel species is not as well developed.

In the future, Tier I monitoring programs are likely to involve more tools from the lower organizational levels as our knowledge and techniques are refined and transferred from the laboratory, and/or experimental field trials to routine field monitoring situations. At the present time the majority of the proposed short-term indices are still in the developmental stages. Since much of the work for these indices has been completed on fishes (aside from marine invertebrate work) a brief introduction to these techniques is provided in ChIV:5.2.7.

Tier II and III biotic monitoring components are directed towards confirming the impact detected in Tier I and identifying the causal agent. Hence, the appropriate Tier II or III biotic monitoring responses are dependent on the nature of the impact detected in Tier I. Hence, it is not possible to state specifically the appropriate Tier II and III components for a generic AMD biotic monitoring program. The range of options for Tier II and III biotic monitoring are briefly outlined in each of the following sections whenever possible.

## 1.0 HYDROLOGY

The source generation and the transport of AMD and its resultant impacts on the aquatic receiving environment are critically linked to the local hydrologic cycle. The generation of AMD is directly influenced by hydrological inputs which are primarily precipitation (i.e., rain or snow, ChIV:1.4). The resultant contaminant pathways are determined by the local hydrologic regime of both surface water and groundwater.

### 1.1 Surface Water Hydrology

#### 1.1.1 Pre-operational Hydrologic Requirements

Surface water hydrology is a significant component of the habitat classification process that should be completed during the environmental characterization phase of any mining operation. The requirements for completing the aquatic habitat classification for a mining operation are addressed in TSN 4.3, with details on the specific hydrological components provided in the following sections.

A strong baseline data collection program providing hydrometric data, for both streams and lakes, is critical to design decisions for developing the mining infrastructure and environmental management. Knowledge of the hydrologic regime of streams, including velocity, stage, discharge and the run-off characteristics of particular watersheds, is required to:

- identify periods with the highest potential for AMD flushing;
- maintain effluent dilution ratios;
- determine reservoir size required to maintain dilution ratios during low flows;
- ensure that bed or bank erosion do not result from the additional discharge associated with the effluent release;
- calculate total loading of various chemical parameters to the environment
- support special studies such as contaminant dispersal (plume delineation); and
- assist with the design of required project infrastructure (e.g., culverts and bridges).

An understanding of lake hydrologic parameters such as volumes (present and historical), mean and seasonal outflow discharges, and historical elevations is necessary to:

- calculate flushing rates which have implications to water quality;
- estimate the nature of contaminant dispersal (plume delineation) and residence time in a receiving waterbody;
- assess suitability for sub-aqueous deposition of potential AMD materials; and
- predict possible changes in lake elevation if water is to be removed from or released into a stream.

#### 1.1.1.1 Regional Surface Hydrology

Regional surface hydrologic data are used to investigate cumulative effects of mining operations in large watersheds and to supply long term information to strengthen local short term flow estimates. Frequency curves for regional flood and low flows should be used to develop preliminary estimates of expected flows in a project area and can be modified to reflect more accurately local conditions once site-specific baseline data become available.

Regional hydrologic data should be obtained from streams which drain watersheds that are similar to the specific area of investigation in terms of watershed area, geology, topography, vegetation, and weather patterns. Hydrologic data for specific streams and lake levels in a region can be obtained from the Water Resources Branch of Environment Canada or from private data related to other projects. Most hydrologic data funded by federal, provincial, or territorial governments are found on the Hydat CD-ROM database which is available for use at many university libraries or, alternatively, may be purchased from Environment Canada.

#### 1.1.1.2 Site-specific Hydrological Requirements

Typically, a lack of streamflow information or the limited time-span of existing flow records in a new project area necessitates comprehensive surface hydrology baseline data collection followed by an ongoing monitoring program. The surface hydrological program should be initiated early in the planning stages so that the natural flow regime can be determined prior to any significant development on the property. The preliminary survey should be designed to provide information critical to the operation of the specific project but, in general, should provide information sufficient to calculate:

- mean discharge for all streams in the study area;
- overall surface water budget of each subwatershed within the project area;
- magnitude and frequency of both high and low flow events for all streams associated with the project;
- flow duration curves for all streams associated with the project;
- water level data for lakes in proximity to mining operations and for several lakes which can be used as controls; and
- mixing zones using mathematical models incorporating predicted plume characteristics and measured hydrological data.

These objectives can be met by:

- The installation of all season continuous stage recorders in streams which are likely to be influenced by mining, milling, dewatering, or waste rock storage. Smaller streams may be monitored with continuous stage recorders during the open water season with occasional instantaneous flow measurements collected during the winter months (if possible).
- The establishment of stage-discharge rating curves at each monitoring location. This requires the installation staff gauges (surveyed to a legal survey monument or an arbitrary benchmark) and completion of an appropriate number (12-20) of cross-sectional instantaneous flow measurements taken over a wide range of flow regimes.
- The installation of staff gauges in any lakes likely to be influenced by dewatering operations or any other water requirements for the mine.

### 1.1.2 Operational Hydrological Requirements

#### 1.1.2.1 Effluent Discharge

Accurate measurement of effluent discharge is required to determine the quantity of effluent released to the environment. Specific discharge monitoring requirements will be detailed in the operating permit. Selection of effluent flow rate monitoring equipment is beyond the scope of this document but is addressed in some detail in Harsham (1995).

### 1.1.2.2 Water Quality Stations

The stream discharge should be measured in conjunction with each water sampling period at each sampling station. This discharge may be determined in one of three ways which are listed here in decreasing order of expense:

1) stage recorder located at the station used in conjunction with its rating curve;
2) instantaneous discharge determined using a current meter; or
3) a staff gauge located at the water sampling site used in conjunction with its rating curve.

### 1.1.2.3 Plume Delineation (Recommended Special Investigation)

It is imperative that designated impact sampling stations, whether water quality or biological, be located within the effluent plume. The modelled plume delineation results should be confirmed during the operational phase when effluent release has commenced. This may be achieved using a conservative constituent of the effluent (e.g. sodium), or preferably, using a tracer dye such as Rhodomine WT. Plume delineation is of special importance if water quality monitoring stations are to be located in large waterbodies, such as lakes with multiple inflows. Under these conditions effluent plumes often exhibit dynamic patterns which shift seasonally due to changes in wind exposure (e.g., ice cover or changes in prevailing winds) and vertically distributed flow patterns (e.g., stratification effects) as illustrated in Figure IV:1.1-1. Plume delineation may not be necessary for small, rapidly mixed waterbodies.

### 1.1.3 Station Location

The location of flow monitoring stations should be assessed according to the following criteria:

- accessibility during all seasons;
- spatial association with areas of influence such as points of effluent release (AMD or milling), water diversion or withdrawal, or control streams;
- a straight reach of the stream, preferably with a uniform cross-section, where both the channel bed and stream banks are stable;
- if possible, the substrate should consist of small, uniform bed material with no vegetation, debris, or other obstacles to impede flow;
- the entire flow should be confined to a single channel at least three stream widths upstream of the measurement section; and

- all other factors being equal, locations immediately downstream of lakes are desirable because relatively warmer water from the lakes may result in channels that have little or no ice cover in winter (i.e., minimal seasonal effect on channel geometry and, therefore, the rating curve).

### 1.1.4 Selection of Equipment

Current meters and stream stage recorders are commercially available in several designs suitable for instantaneous measurements and water level recording. Table IV:1.1-1 summarizes the pros and cons of a number of the commercially available options. Additional information is provided in the accompanying TSN 7.1. Stage recording during the open water season can be accomplished with all designs, but the float and counterweight style cannot be used in the winter months as ice prevents movement of the float. Stage recorders which employ a pressure transducer and data logger system can operate all year and are currently being used at many locations. Other systems using new technology such as acoustics are also being employed with good results in some situations.

## 1.2 Groundwater Hydrology Requirements

Information on the subsurface hydrology of a region is required in order to characterize the quantities, quality and movement of groundwater for the pre-operational, operational, and decommissioning stages of a mine operation.

### 1.2.1 Pre-operational Hydrologic Requirements

A baseline data collection program provides groundwater hydrology data that is important for both the planning of mine infrastructure and the establishment of a groundwater monitoring program. An understanding of the local groundwater regime, including delineation of the piezometric surface and the determination of both direction and approximate rate of flow, is required to:

- indicate and quantify potential groundwater inflows to underground mineworkings and open pits;
- identify areas on the property where groundwater may be very close to the ground surface and thus more prone to AMD effects;
- identify areas on the property where groundwater flow is artesian or could be artesian and thus would contribute to surface flow and AMD considerations associated with surface flow;
- identify significant groundwater recharge areas (if any) on the property upon which the placement of PAG rock should be avoided (e.g., PAG ore stockpiles, PAG waste rock);
- permit the modelling of groundwater contaminant plumes should AMD influence groundwater;
- identify areas where natural ARD occurs: and

- contribute to the data required for the design of project infrastructure (e.g., mine dewatering facilities, tailings disposal facilities).

#### 1.2.1.1 Regional Groundwater Hydrology

The analysis of regional groundwater data is used as a preliminary step in the baseline data collection program. Potential sources of data include exploration/deposit delineation drill cores and logs, geologic maps, and data from other projects in the area. The acquisition and analysis of regional data may provide a groundwater geologist with information on:

- the nature of regional geologic formations;
- the pattern of regional groundwater flow;
- the approximate depths at which groundwater may be encountered;
- the type of geologic materials through which wells may have to be drilled;
- background geochemical concentrations; and
- preliminary approximations of hydraulic conductivity and rates of groundwater flow.

#### 1.2.1.2 Site-specific Groundwater Hydrological Requirements

Detailed information is required about the nature of groundwater hydrology especially in the immediate vicinity of the proposed development. These data will be of value for AMD groundwater monitoring, dewatering and waste management operations. The collection of groundwater data should commence early in the mine planning stage so that baseline conditions can be established before the hydrologic regime is influenced by mining operations (e.g., dewatering operations). A carefully planned groundwater hydrology baseline program will minimize possible duplication, and additional unnecessary cost, of the data needed for AMD and the data required for the planning of mining operations. With reference to AMD, data collection should concentrate on shallow groundwater which is more likely to be influenced by AMD than deep groundwater. Site-specific baseline data should identify:

- the piezometric surface (or water level map) of each groundwater zone;
- the hydraulic conductivity of each zone;
- the direction and rate of flow of groundwater in each zone;
- nonpermeable geologic strata that may be considered natural barriers to groundwater movement (e.g., fine grained clays);

- significant groundwater recharge areas on the property that should be avoided in the planning of infrastructure (e.g., location of waste rock piles, roads, ore stockpiles); and
- regions where groundwater flow is artesian and may contribute to surface hydrology AMD considerations.

These objectives may be met by:

- drilling monitoring wells at locations and depths that are appropriate for the geology of the area;
- establishing the piezometric surface and the direction of flow by measuring the water level in the wells;
- calculating the hydraulic conductivity (via bail or pumping tests) so that the rate of flow may be estimated; and
- using geophysical techniques such as piezo-cone penetration testing (TSN 8.5) and electromagnetic surveys (TSN 8.1-8.4).

Additional information is available on these techniques in the groundwater TSN 14.3.

### 1.2.2 Operational and Decommissioning Requirements

During the operational and decommissioning phases groundwater hydrological monitoring primarily consists of source monitoring as outlined in ChIII2.0. The water levels for the well network should be measured at regular intervals and before any sampling for water quality (ChIV2.2.2.4). The routine monitoring network would benefit from periodic screening along potential groundwater flowpaths using geophysical techniques (TSN 8.0). For information on monitoring groundwater quality, see ChIV2.2.2.

## 1.3 Section Summary

### Surface Water Hydrology

#### *Regional Requirements*

Regional surface hydrologic data are used to investigate cumulative effects of mining operations in large watersheds and to supply long-term information to strengthen local short-term flow estimates. Frequency curves for regional flood and low flows should be used to develop preliminary estimates of expected flows in a project area and can be modified to reflect more accurately local conditions once site-specific baseline data become available.

#### *Site-Specific Pre-operational Requirements*

The preliminary survey provides information critical to the operation of the specific project. This commonly requires a surface **water budget** of each subwatershed within the project area and the calculation of **mean discharge** for streams in the study area. The magnitude and frequency of both **high** and **low flow** events and **flow duration curves** should be determined for all streams directly associated with the project. **Water level** data are required for lakes in proximity to mining operations and for several lakes which can be used as controls. By incorporating these hydrological measurements with predicted plume characteristics **mixing zones** can be mathematically modelled.

#### *Operational Requirements*

Accurate measurements of end-of-pipe **effluent discharge** are required to determine the quantity of contaminants released in the effluent to the environment. Discharge should also be measured at each **water sampling station** for every sampling interval.

Once effluent release has commenced the modelled plume delineation results should be empirically confirmed. **Plume delineation** is of special importance if water quality monitoring stations are to be located in large waterbodies, such as lakes with multiple inflows.

### Groundwater Hydrology

Information on the subsurface hydrology of a region is required to characterize the quantities, quality, and movement of groundwater for the pre-operational, operational, and decommissioning stages of a mine operation.

### *Regional Pre-operational Requirements*

The analysis of regional information is used as a preliminary step in the baseline data collection program. Information requirements include the nature of regional geologic formations, the depth, flow patterns, preliminary rates of hydraulic conductivity, background geochemical concentrations for the regional groundwater systems, and the type of geological material through which wells may have to be drilled.

### *Site-Specific Requirements*

Detailed site-specific groundwater hydrology is required to assist in designing dewatering and waste management plans, and the AMD groundwater monitoring program. For AMD purposes, data collection should concentrate on shallow groundwater systems which are more likely to be influenced by AMD than deep groundwater. Site-specific baseline data should identify:

- the piezometric surface (or water level map) of each groundwater zone;
- the hydraulic conductivity of each zone;
- the direction and rate of flow of groundwater in each zone;
- nonpermeable geologic strata that may be considered natural barriers to groundwater movement (e.g., fine grained clays);
- significant groundwater recharge areas on the property that should be avoided in the planning of infrastructure (e.g., location of waste rock piles, roads, ore stockpiles); and
- regions where groundwater flow is artesian and may contribute to surface hydrology AMD considerations.

### *Operational Requirements*

During the operational and decommissioning phases groundwater hydrological monitoring is incorporated into the source monitoring as outlined in ChIII2.0. The routine monitoring network would also benefit from periodic geophysical screening of potential groundwater flowpaths.

## 1.4 Section Bibliography

**Discharge Measurements**

Harsham 1995

Terzi 1981

**Field Hydrology**

Brassington 1990

**Hydrology Texts**

Dunne and Leopold 1978

Gordon et al. 1992

Richards 1982

Shaw 1983

**Field Methods**

Brassington 1990

Terzi 1981

**Stream Hydrology**

Harrelson et al. 1994

## 2.0 WATER QUALITY MONITORING

The core component of any AMD monitoring programme is the water quality component. Water monitoring may be compartmentalized into two basic classifications:

1) surface waters (ChIV:2.1), both flowing and standing; and
2) subsurface water (ChIV:2.2).

General characteristics of these two water compartments, and concerns specific to monitoring AMD in these systems are provided in the subsections to follow.

## 2.1 Surface Water Quality

Monitoring of surface water quality for AMD purposes can be subdivided into **Source** monitoring and **Receiving Environment** monitoring. Monitoring considerations specific to Source monitoring were addressed in ChIII:2.0 with the reader directed to the pertinent portions in this section which are applicable to both source and receiving environment monitoring.

### 2.1.1 Lakes

Lakes generally buffer and dampen seasonal contaminant inputs through dilution, and possess a number of seasonal physical, chemical, and biological characteristics which are strongly influenced by the thermal characteristics of water. These factors must be considered when sampling in lakes and are briefly discussed below.

In response to seasonal temperature change, lakes in temperate climates often undergo an annual cycle of stratification and destratification. Stable stratification usually occurs in waterbodies with depths greater than 10 m (Wetzel 1983). During the warm weather months lakes absorb energy from the sun and air, resulting in increasing water temperatures. During cold weather periods the lake loses heat energy to the air resulting in a net decrease in water temperatures. Since water density is influenced by temperature, seasonal stratification occurs when warm, less dense waters near the surface (epilimnion) overlie colder, denser waters (hypolimnion). These two layers are separated by a layer in which in which the temperature decreases rapidly with depth (metalimnion) (Figure IV:2.1-1a,b). The metalimnion acts as a barrier to mixing between the epilimnion and the hypolimnion. This thermal stratification may be associated with dissolved oxygen and chemical stratification, which has significant implications for contaminant and natural background water quality.

Stratification may break down at certain times of the year resulting in the mixing of waters from the epilimnion and hypolimnion. Spring turnover occurs when the surface water warms and becomes denser (maximum density of water is near 4°C) moving downward through the water column displacing the lighter, cooler waters below. Autumn turnover occurs as warm water in the epilimnion cools to reach maximum density and sinks in a similar manner as during the spring turnover. Winds aid turnover events during both spring and fall. During these turnover events the waters of the lake mix and the concentrations of chemical constituents become more uniform throughout the water column.

In shallow lakes wind energy and, low water residence times tend to prevent stratification, though stratification may occur during sunny, warm, calm weather.

In both shallow and deep lakes, stratification can occur if inflow waters differ significantly in temperature from the resident waters. Cold water inflows from tributaries or underground aquifers may flow along the bottom of the lake and warm water effluent inputs may also stratify for a period.

### 2.1.2 Rivers and Streams

Lotic systems are flowing bodies of water which tend to exhibit fluctuations in discharge and water quality in response to short-term seasonal variations in precipitation inputs (i.e. rainfall and snow-melt). The seasonal influences on discharge are evident in the annual hydrographs shown in Figures II:1.4-1 and II:1.4-2 for a coastal mine and an inland mine, respectively. This seasonal pattern is reflected in increases or decreases in AMD flushing and dilution rates. As discussed in ChII:1.4, flushing events after periods of AMD accumulation at the source initially result in increased concentrations of AMD associated contaminants in the receiving waters. This may soon be masked due to the dilution of AMD contaminants with additional run-off from the sub-basin and the hysterisis effect.

Seasonal patterns are associated with other chemical and physical characteristics which may influence environmental impacts associated with AMD. For example, seasonal patterns in alkalinity result in periods of limited acid buffering capacity for streams. Seasonal periods of low flow which may correspond with periods of low oxygen concentration (e.g., high summer temperatures with low precipitation) can result in marginal dilution of contaminants during a period already stressful to many organisms. Such background water quality patterns need to be understood to predict periods of maximum AMD sensitivity to minimize contaminant effects in the receiving environment.

### 2.1.3 Environmental Characterization

The objective is to gain a general characterization of the water quality and limnological characteristics of potential impact and reference waterbodies. This information will be used to identify potential control/reference and receiving environment monitoring stations for upstream and downstream waterbodies, which will be resampled during the completion of the intensive baseline study.

### 2.1.3.1 Lakes

#### *Limnological Data*

The following are recommended minimum requirements for lakes which may be influenced by the mining operation, as well as for one or more potential reference lakes. It is advisable to complete these procedures for all minesites independent of their AMD potential.

- complete **temperature** profiles to determine the seasonal thermal stratification pattern for the lakes;
  -identify to what extent thermal stratification occurs in the winter or summer and whether spring and autumn turnover (mixing) occurs;
- determine the **dissolved oxygen**, **conductivity**, and **pH** profiles which accompany the temperature profiles; and
- determine **Secchi disk** transparency
  -roughly identifies the depth at which light penetration is approximately 10% of that at the water surface.

The limnological information can be determined using field meters. Data for vertical profiles should be collected every 0.5 m for shallow water bodies (e.g. ≤ 3 m) and every 1 m for deeper waterbodies (≥ 3 m). This limnological data should also be collected in support of all additional aquatic ecological investigations (e.g., sediment, benthic macroinvertebrate, etc.).

#### *Water Chemistry*

Seasonal water chemistry should also be collected. Stratified waterbodies should be vertically sampled. Usually a minimum of five vertical sampling points are recommended:

1) surface;
2) immediately above the thermocline;
3) immediately below the thermocline;
4) mid-hypolimnion; and
5) 0.5 to 1 m above the sediment-water interface (IHD-WHO 1978).

If budget restrictions are a concern, this may be decreased to three samples, one from each of the epilimnion, metalimnion and the hypolimnion.

During this environmental characterization phase a broad range of water chemistry parameters should be determined. These would consist of:

Buffering Capacity
- acidity
- alkalinity

Physical Properties
- total dissolved solids (TDS)
- total suspended solids (TSS)

Major Ions
- bicarbonates
- chlorides
- sulphates
- calcium
- magnesium
- sodium
- total hardness

Nutrients
- organic carbon
- nitrogen (ammonia, nitrite, nitrate)
- phosphorous (total and organic)

Metals
- ICAP Scan

Site/Process Specific Parameters
- e.g. radionuclides

At this stage the metals can be analyzed using a standard ICAP scan. This will serve to identify those metals which will require analytical methods with lower detection limits during completion of the intensive baseline. Table IV:2.1-1 lists detection limits, for various methods, for trace elements in water.

### 2.1.3.2 Rivers and Streams

A similar general water quality program should be completed for the major rivers or streams entering and leaving the area of the proposed mine site.

- Complete limnological profiles in systems ≥ 1 m deep using profile rules as for lakes. In shallow streams use mid-depth measurements.
- Collect a seasonal water chemistry sample upstream and downstream of any river or stream which may receives drainage or point source releases from the mining project.

- collect samples from mid-depth of the main channel or;
- collect vertically stratified samples if limnological profiles exhibit stratification.

#### 2.1.3.3 Investigating Background ARD

In addition to the above generic water quality characterization program, an ARD water quality investigation should be completed for the streams and rivers in the region. A specific investigation for natural ARD is not recommended for lakes as ARD inputs tend to be difficult to detect due to dilution. The objective is to document the watersheds' background concentration of ARD contaminants from sources other than the proposed mine. Weathering of natural sulphide-bearing outcrops, or materials exposed by rock slides or road building can produce ARD contaminants. Background AMD contaminants can also be present from other operational or abandoned mines in the same watershed. Without an appropriate pre-operational baseline it is not possible to separate these sources from possible future inputs from the proposed mine.

Determination of background ARD/AMD will require a short-term investigation during the predicted peak acid release period(s) to the environment. The following points should be considered:

- determine potential peak release periods from the regional hydrologic data (ChIV:1.1);
- sample all major rivers and streams associated with the project;
- sample smaller tributaries based on regional surface geology to define source areas - e.g., tributaries associated with surface run-off from sulphide bearing outcrops or anthropogenically exposed (e.g., forestry roads, other mines) sulphide deposits; and
- sample a minimum of weekly during the predicted peak period.

The standard Tier I water quality parameters of pH, acidity, alkalinity, specific conductance, and sulphates should be determined. An ICAP trace metal scan or analyses for specific metals based on the surface geology may be included. The decision to analyze for trace metals or additional site-specific parameters may be restricted to those sites showing evidence of possible acid drainage based on the Tier I parameter results.

Should there be evidence of acidic streams influenced by ARD, the local surface geology should be mapped in relation to stream water quality, and the significant ARD contributors identified. At this point care should be taken to identify possible AMD contributions from exploration activities. An example is provided in Figure IV:2.1-2, of the MacMillan Pass in east-central Yukon, where acidic streams with elevated metal concentrations were found to be associated with specific sulphide-bearing outcrops (Kwong and Whitley 1992) more common to the east side of the MacMillan River. The predominant outcrops on the west side possessed greater buffering capacity which neutralized ARD. Such information collected during the pre-operational phase supports waste management decisions (ChII:2.2.1.3 and III:1.1.1), which, in this example, would involve locating potential AMD sources (e.g., mills, piles, etc.) on the west side of the river to take advantage of the natural buffering capacities (Kwong and Whitley 1992; MEND 1.32.1 1993).

### 2.1.4 Development of Operational Monitoring Program

Water quality monitoring programs for the receiving environment are subdivided into three program types:

1) **Trend Monitoring**;
2) **Monitoring Total Loadings to Lakes**; and
3) **Monitoring of Sensitive Habitat/Species**.

#### 2.1.4.1 Trend Monitoring Programs

In the mining industry **trend monitoring** is the traditional and most common form of water quality monitoring used in the receiving environment. Monitoring programs designed for trend analysis are not suitable for early detection of AMD associated impacts, though they can provide evidence of long-term trends when an extensive database has accumulated (e.g. ≥ 10 years). Hence, trend monitoring systems are valuable during the decommissioning phase. Water quality trend monitoring is usually a monitoring component completed at mine sites independent of the AMD potential of the mine. The standard protocols involve the collection of single samples using fixed sampling intervals.

***Station Locations***

Receiving environment monitoring programs should always be developed using a BACI experimental design and whenever possible a multi-control design. Specific station locations should be determined as follows:

- Identify potential contaminant pathways for possible point source (e.g. treated AMD facility) and non-point source (e.g. surface or overland drainage) AMD. Extend Source pathways (ChIII:2.0) to the receiving environment incorporating any sensitive or valued habitats identified during the biotic environmental characterization study .

- Control/reference stations should be located upstream of the identified potential contaminant inflow points.
- The first potentially influenced receiving environment station (near field) is located at the point where complete mixing of the source discharge with the receiving flows (based on best estimate or preliminary plume modelling) is predicted.
- A final downstream station should be located beyond the predicted point of potential environmental effects (far field).
- Additional stations are required between the near and far field stations if:
  - a sensitive habitat is identified (e.g. fish spawning habitat, see ChIV7.1.4.3); and
  - there is the possibility of additional contaminant influx from other anthropogenic activities in the watershed.

The appropriate placement of these stations in fully mixed zones (effluent and natural waters) should be confirmed as soon as effluent or contaminant release has commenced. This is required if unreplicated, single samples are to be justified in the sampling program. This can be accomplished by completing a tracer dye plume investigation or by collecting replicated samples laterally along a cross-section of the sampling station. These data will identify the within-station variance and hence an error estimate for the reliability of single samples. If a station is found to be incompletely mixed, the number of replicate samples required to adequately represent the mean at the station can be determined using power analysis, or the station may be relocated.

Lakes are not ideally suited for trend monitoring due to the potentially high seasonal variability of effluent plume locations (e.g., Figure IV:1.1-1). Investigations on total contaminant loadings (see ChIV2.1.4.2) are often more accurate and cost efficient compared to the expence assocaited with the number of stations (vertically and laterally) required to accurately reflect water quality within a lake. If lakes are to be incorporated in the trend monitoring program the sampling station should be located in the deepest point of the lake to support any sediment monitoring program (ChIV3.3.1). In large lakes additional stations should be located at the inflow(s) and outflow(s) of the lake.

Vertical subsampling may be required due to natural seasonal stratification events and incomplete vertical plume mixing. If limnological profiles exhibit stratification, then vertical subsampling of the water column at a station should be completed using the protocol described in ChIV2.1.3.1. In most cases the conductivity profile should serve to identify whether complete vertical mixing of treated effluents is occurring, and hence, identify whether vertical sampling is required. To confirm this a short term investigative program (e.g. one full sample season) consisting of vertically stratified sampling with replications within each vertical stratum should be considered once the release of effluents has commenced.

### *Sampling Frequency and Parameters*

- Collect the water samples on a monthly basis, budget permitting. Quarterly sampling, timed to incorporate turnover events in the drainage, may be considered.
- The parameter list need not be as detailed as the environmental characterization list. Water quality parameters of interest will be site- and ore-type specific and must at a minimum incorporate those required by the site's regulatory body.
- If long-term trend monitoring of AMD components is also an objective of the program then the following parameters should be incorporated if not already in the parameter list:
    - pH
    - total dissolved solids / conductivity
    - acidity
    - alkalinity
    - sulphate
    - redox potential
    - discharge
    - temperature
    - metals of concern identified in the predictive AMD assessment

- Parameters found to be below detection limits in the environmental characterization studies should be reviewed to determine whether more sensitive analytical tests are required.

### *Intensive Baseline for Trend Monitoring*

The purpose of the water quality intensive baseline is to provide the temporal control data for future trend analysis.

- Locate stations at the same sites as selected for the future monitoring program.
- Locate additional stations if there is the possibility of project design changes relating to points of release.
- The parameter list should be broader than that required for the operational monitoring program. Parameters should include those listed for the environmental characterization and the above mentioned AMD monitoring parameters.

- Determine suitable analytical detection levels. If a test's detection level is near potential risk levels, e.g., the concentrations recommended by the Canadian Water Quality Guidelines for the Protection of Aquatic Life (1987), then more sensitive analytical techniques should be considered.

***Operational Trend Monitoring***

The operational monitoring program should be executed as designed, incorporating any fine- tuning identified after completion of the intensive baseline. The first year of the operating program may require additional sampling (e.g., verticle and/or horizontal) to confirm mixing.

### 2.1.4.2 Large Waterbodies and Loadings

As mentioned earlier in this section, lakes can be poor water-quality monitoring stations due to the dynamics relating to plume positioning in both the horizontal (e.g., wind, currents) and vertical (e.g., stratification) planes. Since lakes tend to act as sinks for contaminants, the most productive approach to lake monitoring is likely to involve monitoring water-quality contaminant loads into and discharged out of a lake in order to determine the contaminant load retained in the physical (e.g., sediments) and biological (e.g, benthic macroinvertebrates, fish) sinks of the lake. To improve the cost efficiency of such a program, parameter selection should involve a restricted list (one to two items) of identified contaminant(s) of concern. At the same time there should be analyses to determine the biologically active and inactive composition of the contaminants in the discharge and the various sinks.

Such an approach would require a water quality program designed to monitor **mean** inflow and outflow concentrations as described in ChIII:2.5.2.2. Hence, due to budget constraints it would likely only be cost effective for near field lakes recognised as at moderate to high risk from AMD. The water-quality program would subsequently be supported by monitoring programs of the physical (e.g., sediments) and biological (e.g., macroinvertebrates, fish) sinks within the lake. These sinks would be monitored as detailed in the following sections.

For example, assume AMD drainage containing elevated copper concentrations discharges into a lake. Water quality stations would be established at the inflow and outflow to determine annual mean copper loadings (total and dissolved), incorporating a stratified sample design (ChIII:2.5.2.2). The dominant physical and biological sinks would also be monitored (ChIV:3.0 - 6.0). Such a program would provide an annual copper budget for the lake, with information on the accumulation of copper in the lake as a whole, the compartmentalization of this copper within the sub-components (physical and biological) of the lake, and the proportion which was bioavailable. Risk management decisions can be made based on the evidence of contaminant movement or storage between abiotic and biotic compartments.

### 2.1.4.3 Monitoring for Sensitive Habitats/Species

If **sensitive habitats and/or species** (hereafter referred to as habitats) were identified by the environmental characterization, and alternative mining options (e.g. different effluent release points, etc.) were not feasible, then a water quality monitoring program may be developed specifically for that habitat. Alternatively, the unforeseen development of an AMD discharge into a sensitive habitat may require specific monitoring or compliance requirements. Such a scenario is present at the Mt. Washington AMD site where copper contaminated AMD drains into salmon spawning and rearing habitat (BC Environment 1995).

Once again, the type of monitoring program required would depend on the degree of concern and whether peak or mean contaminant concentrations were of interest (see ChIII:2.5.1.2). These monitoring programs should also have a restricted parameter list based on identification of the AMD trace metal of most concern and incorporating total and dissolved concentrations.

If the sensitive organism(s) is exposed or present at that site for only a specific short time interval, such as fish eggs deposited at a site which is not a rearing habitat, then monitoring of peak concentrations during this short time interval would likely be the most cost efficient design. Should the site also be rearing habitat for fry and fingerlings, then peak monitoring for this extended time period may not be cost effective[10]. This situation may be best monitored using a stratified monitoring design to determine mean concentrations. The mean concentrations could be statistically analyzed across years to determine whether the sensitive habitat was being exposed to stable or increasing AMD concentrations, and thereby provide managers with suitable response time to alleviate potential fisheries concerns which can have important regulatory ramifications.

An example of water-quality objectives designed for the protection of fish spawning and rearing activities was proposed in 1995 for the Tsolum River receiving AMD from the Mount Washington

---

[10] Cost effectiveness would have to be assessed relative to the potential risk associated with the specific AMD concern. If AMD concern was HIGH peak monitoring may be justified.

Mine in British Columbia. This water quality objective recognizes the seasonality of AMD, proposing that weekly sampling be completed between 15 April to 30 June and 15 September to 30 November, respectively, with an objective of 0.007 mg/L (30 day average) for copper with a maximum concentration of 0.011 mg/L (BC Environment 1995).

An operator of an active mine facing such a water quality requirement should consider establishing an upstream diagnostic station to provide an early warning of potential compliance difficulties. This would require a monitoring program designed to track peak contaminant concentrations as proposed in ChIII:2.5.2.2.

### 2.1.5 Decommissioning Monitoring

A pre-decommissioning baseline program should be completed in the receiving environment which would be similar to the pre-operational baseline discussed in ChIV:2.1.3. The water quality monitoring program during the decommissioning phase should involve the continuation (possibly scaled down) of the operational trend monitoring program. This program will now be of more value due to the accumulation of data over the operational period of the mine.

Additional monitoring requirements will depend on a risk assessment completed using the results of the long-term AMD modelling for the potential sources and control structures. The continuation of a scaled down version of the operational monitoring program in the receiving environment, in combination with the source decommissioning water-quality program, should meet the environmental requirements for most mine sites. However, contingency plans should be in place to initiate receiving environment investigations in response to evidence of increased AMD concerns arising from the source monitoring programs. The pre-decommissioning baseline should be designed to support these contingency plans.

### 2.1.6 Sampling Equipment and Protocols

Water sampling equipment for the receiving environment generally consists of simple hand grab samples (TSN 14.1). Hand grab samples should be collected using the following procedures:

- Determine standard limnological measurements *in situ*.
  - field pH, conductivity, dissolved oxygen, and temperature
- Use correct sample container (Table IV:2.1-2).
- Always approach sampling sites by facing and moving upstream. This prevents contamination due to disturbance of the water column or from resuspension of sediments into the water column.
- Rinse sample jars two to three times with the water to be sampled.
- Surface samples should be collected at least 0.5 m below the surface or above the sediment to prevent collection of floating solids.
- Samples should be preserved and stored in the manner appropriate for the parameter to be measured (Table IV:2.1-2).

Samples from depth are most commonly sampled using a grab sampler triggered shut by a messenger, such as Van Dorn or Kemmerer samplers. Another option is a pump sampler, which pumps water up from the desired sample depth.

### 2.1.7 Quality Assurance and Quality Control

The standard generic QA/QC procedures outlined in ChII:2.6.1 should be followed. Samples should be clearly labelled as WATER samples and identified as filtered or unfiltered. Additional labelling should include:

- site and sample identification;
- date and time of collection;
- preservative if any; and
- identification of sampler or sample team.

Additional documentation which should be provided with the field notes is shown in Table IV:2.1-3. For additional discussions on field QA/QC protocols, see BC AMD (1992). The use of Quality Assurance samples, as detailed in (Table II:2.6-1) will depend on the perceived sensitivity of the monitoring program to sampling and analytical errors or biases. The appropriate quality control procedures for the compliance program will be dictated in the permit. An example of an action flow chart is provided in Figure IV:2.1-3, showing the steps to be initiated if there is evidence of anomalous water quality results.

## 2.2 Subsurface Water

Subsurface water has been discussed previously in this document, with ChIII:2.0 addressing basic source monitoring concerns and ChIV:1.2 detailing groundwater hydrological considerations. This section focuses on sampling protocols and considerations for subsurface water quality investigations.

### 2.2.1 Subsurface Water and AMD

In temperate regions of the world subsurface water constitutes the largest source of storage for the hydrologic cycle. The regolith, the loose and discontinuous blanket of decayed rock debris overlying solid bedrock, can act as a storage medium for water or can transmit water vertically or horizontally to bedrock storage sites (Driscoll 1989). Water which has infiltrated the regolith is often classified into one of two simplified zones: the vadose zone or the phreatic zone (Figure IV:2.2-1). The vadose zone is bounded by the soil-atmosphere interface and the groundwater table and may be further subdivided into three layers:

1) the upper soil water;
2) the intermediate vadose zone; and
3) the capillary fringe.

The phreatic zone is sometimes incorrectly referred to as the zone of saturation[11]. A more accurate definition defines the phreatic zone as the zone containing water that freely enters wells under both confined and unconfined conditions. For the purposes of this document, water in this zone is referred to as groundwater.

The lower boundary of the groundwater zone is marked by a transition zone consisting of water in unconnected pores which cannot flow to wells. Below this zone, water is chemically combined in the rock forming minerals (Driscoll 1989).

---

[11] This is a misleading definition in fine grained sediments since the lower layer of the capillary fringe may also be saturated (Driscoll 1989).

Contamination of groundwater with AMD is greatly influenced by the length of the pathway and the neutralization potential of the matrices surrounding the pathway. However, unless AMD directly enters the groundwater, as can occur in flooded mine workings, contaminated water will flow through and be influenced by the vadose zone. In addition, groundwater movement is extremely slow relative to surface flow allowing plenty of time for contaminated groundwater to be influenced by the surrounding geochemistry. As a result groundwater presents a long, slow AMD pathway which often acts to neutralize acid mine drainage.

### 2.2.2 Groundwater Monitoring

#### 2.2.2.1 Environmental Characterization

In addition to the groundwater **hydrological** requirements discussed in ChIV:1.2, the pre-operational groundwater **quality** should be determined. Potential groundwater contaminant pathways should be developed to anticipate the design options that may be included in the final mine plan. At a minimum, pre-operational groundwater quality should be determined up-gradient and down-gradient for each groundwater stratum with the potential to be influenced by the mining operation. The groundwater investigative network developed to meet mine design and engineering requirements (e.g., dewatering plans) will likely already meet the needs of the groundwater quality requirements at this stage of the project (ChII:1.2.1).

The recommended US EPA (1991b) parameter list for the pre-operational stage is shown below:

- standard groundwater chemical parameters
  - pH, specific conductance, TOC, TOX, alkalinity, TDS, Eh, $Cl^-$, $NO_3^-$, $SO_4^{2-}$, $PO_4^{3-}$, $SiO_2$, $Na^+$, $K^+$, $Ca^{2+}$, $Mg^{2+}$, $NH_4^+$, Fe, Mn;
- additional potential impact assessment parameters
  - B, Zn, Cd, Cu, Pb, Cr, Ni, Ag, Hg, As, Sb, Se, Be;
- additional site-specific parameters not mentioned above; and
- water-quality hydrologic parameters
  (See ChIV:1.2 for complete hydrologic requirements.)
  - water level, hydraulic conductivity.

These are recommended as baseline parameters when the specific potential contaminants are not defined; e.g., when there is the potential of both organic and metal contamination. The organic components may be dropped if the operator is convinced that AMD is the only potential contaminant and that contamination from organic compounds (e.g., sewage, organophosphates, solvents, etc.) is unlikely. The conservative design would measure all of the parameters for the baseline and then refine the operational parameter list.

### 2.2.2.2 Monitoring Program Development

#### *Well Locations*

The groundwater monitoring program consists of Source and Receiving environment monitoring. Due to the slow movement rates (relative to surface waters) groundwater monitoring tends to be concentrated near to or within the source (ChIII:2.0). If the potential AMD sources are limited and the contaminant pathways well defined, then the minimum receiving environment requirement of an up-gradient (control) and one down-gradient (potential impact) well per station is often adequate. This assumes that adequate source and near source monitoring networks have been established. Additional wells may be added to the receiving environment network should the source and near source wells indicate AMD migration. If there are a number of potential contaminant sources, then the well network should be designed to be able to identify between the potential sources. Site-specific concerns due to short contaminant pathways, such as a groundwater system in close proximity to surface flows, would require a denser monitoring network and possibly vadose zone monitoring (ChIV:2.2.3).

#### *Preliminary Investigation / Intensive Baseline*

Once all of the wells for the monitoring program have been located and drilled (ChIV:2.2.2.2), an **intensive baseline** monitoring program should be completed. Upon completion of the environmental characterization and the intensive baseline, the chemical and hydrogeologic characteristics of the property and each well (ChIV:1.2.1.1) should be understood (e.g., spatial distribution, hydraulic conductivity, etc.). This can be achieved by selecting from the various procedures discussed in the groundwater TSN (14.3). The intensive baseline program should involve:

- collecting water samples from each water quality station on a monthly basis for one year;
- completing an investigation to identify the instantaneous variation by collecting three to five replicates at each station for a minimum of one sampling event;

- determining the required replication for operational monitoring program by completing a power analysis on these data; and
- collecting samples should be collected using the procedures discussed in ChIV:2.2.2.3. The same parameter list as detailed for the environmental characterization (ChIV:2.2.2.1) should be used.

The data from this intensive baseline should provide the information needed to develop the operational monitoring program.

***Operational Monitoring Program***

The groundwater flow rates and water quality data from the preliminary survey should be investigated for any evidence of seasonal patterns and correlations with surface water discharges.

- Sampling frequency for the monitoring program should be based on the degree of autocorrelation in the data set from the preliminary survey. Groundwater tends to exhibit a high level of autocorrelation; hence, sampling intervals can likely be separated by a number of months.
- The parameter list for the operational program should be reduced to the standard hydrologic characteristics, Tier I AMD indicators for water quality and to the site-specific parameters as determined in the risk assessment modelling. Due to the lower sampling frequency, it may be cost effective to also measure Tier II AMD parameters each sampling event.

### 2.2.2.3 Monitoring Well Design Considerations

The construction of wells (e.g., diameter, screen type, casing material) is site-specific and should be determined by a hydrogeologist. The appropriate drilling method is dependent on the subsurface characteristics and the required depth. The various drilling options are shown in Table IV:2.2-1.

There are two general types of groundwater samplers, single or standpipe wells, well clusters, and multiple port piezometers, which are both shown in Figure IV:2.2-2 and described further in the groundwater (TSN 14.3) Care must be taken when drilling and constructing the well as these processes can themselves impact on the water-quality. Proper well development is essential if a well is to be used for water quality purposes. The use of improper materials in well construction may result in inaccuracies in baseline chemical analyses of the groundwater or in wells that are unacceptable for operational monitoring purposes (See TSN 14.3.3 for well design).

### 2.2.2.4 Basic Well Sampling Protocols

An outline of a generalized groundwater sampling protocol is shown in Table IV:2.2-2, with Figure IV:2.2-3 providing additional supportive information. The protocols will be briefly addressed.

#### *Water Level Measurements*

Prior to purging or sample collection, the water level in the well should be determined. This serves a number of purposes:

- to estimate purging requirements;
- low levels may indicate the influence of the cone of depression of production wells;
- increased water levels may indicate recharge events; and
- historical well levels may be used to identify seasonal patterns.

The water level may be determined in one of three ways depending on the depth of the well, level of accuracy required, and the expenditures allocated to this component:

1) steel measuring tape;
2) electric drop line; and
3) pressure transducer/digital data logger.

The advantages and disadvantages of each method are summarized in Table IV:2.2-3.

#### *Purging*

Purging of the well insures that the water samples reflect the present water quality in the strata and not in the well shaft. The purging volume should be calculated based on the hydraulic characteristics of the well. However, it is recommended that this estimate not be used for the actual purging. The most accurate means of purging a well is to continue the purging process until the well-purging parameters (pH, conductivity, temperature, redox) stabilize. Should the actual and predicted purging volume greatly differ, then a number of possible causes should be explored (U.S. EPA 1991b):

- the well may have been sampling the outer margin of a contaminant plume at the beginning of the purging process and have drawn in uncontaminated water from outside the plume by purging;

- the hydraulic properties of the well may have changed indicating a need for well rehabilitation or maintenance;
- the device used for determining the purging parameters may need servicing; and/or
- the water in the well shaft may have been contaminated by the addition of clean water or a contaminate in order to bias the results.

### *Sample Collection and Handling*

A simple review of the pros and cons of a wide range of monitoring well-sampling equipment is provided in Table IV:2.2-4. The selection of a sampling pump is primarily based on a trade-off between the ability to rapidly purge the well of stagnant water (e.g., L/min) and also collect samples at low flow rates (e.g., 100 ml/min) in order to minimize turbulence and degassing (U.S. EPA 1991b).

The collection of the water-quality samples may commence when the well parameters have stabilized over two successive well volumes (U.S. EPA 1991b). The order in which samples are collected is dependent on the sensitivity of the parameter to the required handling process. The flow chart in Figure IV:2.2-3 describes the recommended sampling priorities for a broad range of monitoring parameters. If the well is solely for monitoring AMD constituents, then the organic parameters in the figure are not relevant.

Preservation techniques and recommended holding times are similar to those outlined in Table IV:2.1-1.

### *Quality Assurance and Quality Control*

The standard generic QA/QC procedures outlined in ChII:2.6.1 should be followed. Samples should be clearly labelled as GROUNDWATER samples and identified as filtered or unfiltered. Additional labelling should include:

- well identification code;
- date and time of collection;
- preservative if any; and
- identification of sampler or sample team.

Additional documentation which should be provided with the field notes is shown in Table IV:2.1-2. For additional discussions on field QC requirements, see BC AMD (1992). The use of Quality Control samples, as detailed in (Table II:2.6-1) will depend on the perceived sensitivity of the monitoring program to sampling and analytical errors or biases. The appropriate quality control procedures for any compliance groundwater wells will be dictated in the permit.

### 2.2.3 Monitoring the Vadose Zone

Vadose-zone monitoring detects contaminants in the vadose zone prior to their entry into the groundwater. The most common form of vadose-zone monitoring for the mining industry involves the monitoring of pore liquid. Pore liquid may be collected from the unsaturated zone directly, using suction samplers or indirectly by core sampling. Core sampling has two significant drawbacks compared to suction sampling. Core sampling is destructive; therefore, it is not possible to collect repeated measurements from the same specific site. It also requires higher atmospheric pressures to extract the water, which results in solute concentrations considered less representative of *in situ* pore liquid compared to water retrieved with suction samplers (Zabowski and Ugolinin 1990).

The most common types of suction samplers are porous samplers often referred to as suction lysimeters (TSN 14.2.4). Suction lysimeters are used in the mining industry to detect leakage problems early enough to allow the initiation of remedial action before the contaminant becomes widespread in the vadose zone or enters the groundwater. Figure IV:2.2-4 provides two case examples where suction lysimeters were installed as diagnostic monitoring tools in the mining industry.

For a review of the principles and practices in vadose-zone monitoring the reader is directed to the “Handbook of Vadose Zone Characterization and Monitoring” (Wilson et al. 1995). Additional references are provided in the section bibliography.

## 2.3 Section Summary

Standing waterbodies (lentic) such as lakes generally buffer and dampen seasonal contaminant inputs through dilution, and possess a number of seasonal physical, chemical, and biological characteristics which are strongly influenced by the thermal characteristics of water. Lotic systems are flowing bodies of water which tend to exhibit fluctuations in discharge and water quality in response to short-term seasonal variations in precipitation inputs (i.e. rainfall and snow-melt). These factors must be considered when developing monitoring programs for surface waters.

## Environmental Characterization

The objective is to characterize the water quality and limnological features of potential impact and reference waterbodies. This information will be used to identify potential control/reference and receiving environment monitoring stations.

### *Limnology and Water Quality*

Temperature, dissolved oxygen, specific conductance, and pH profiles along with measurement of the rate of light extinction (Secchi disk or light meter) should be completed for each potential exposure and reference lake. The same limnological data should be collected for the major rivers or streams entering and leaving the area of the proposed mine site as well as in support of all additional aquatic ecological investigations (e.g., sediment, benthic macroinvertebrate, etc.).

At a minimum, water chemistry should be determined on a seasonal basis with stratified waterbodies vertically sampled. A broad range of parameters is required including major and minor ions, physical properties, nutrients, elements, and site/process specific parameters.

### *Investigating Background ARD*

An ARD water quality investigation should be completed for the streams and rivers in the region. The objective is to document the background concentration of ARD contaminants from sources other than the proposed mine. This requires a short-term investigation during the predicted peak acid release period(s) to the environment.

## Operational Monitoring Program(s)

### *Trend Monitoring*

Trend-monitoring in the mining industry is the traditional and most common form of water-quality monitoring used in the receiving environment. Trend-monitoring study designs are not suitable for early detection of AMD associated impacts, though they can provide evidence of long-term trends when an extensive database has accumulated (e.g. ≥ 10 years).

### *Large Waterbodies and Loadings*

Lakes often act as sinks for contaminants; hence, the most productive AMD lake monitoring design is one that addresses concerns related to **contaminant loadings**.
The objective is to determine the contaminant load into and out of the lake, and the contaminant load retained in the physical (e.g., sediments) and biological (e.g, benthic macroinvertebrates, fish) sinks of the lake. This requires the monitoring of **mean** inflow and outflow concentrations and discharges, combined with monitoring programs of the physical and biological sinks within the lake.

### *Monitoring Sensitive Habitats/Species*

The potential or actual discharge of AMD into a sensitive habitat may require specific monitoring or compliance requirements. The study design will depend on the degree of concern and whether peak or mean contaminant concentrations are of interest.

## Decommissioning Monitoring

A pre-decommissioning baseline program should be completed in the receiving environment. The operational trend-monitoring program should be continued, though the required number of stations and sampling intensity may be reassessed. Additional monitoring requirements will depend on a risk assessment completed using the results of the long-term AMD modelling for the potential sources and control structures.

## Sampling Equipment and Protocols

Water-sampling equipment for the receiving environment generally consists of simple grab samples. Samples from depth are most commonly collected using a grab sampler triggered closed by a messenger. Pump samplers which draw water from the desired sample depth are also an option. Integrated samplers such as tygon tubes may be used when stratification is not a concern. Standard QA/QC procedures should be followed. The use of Quality Assurance samples will depend on the perceived sensitivity of the monitoring program to sampling and analytical errors or biases.

## Subsurface Water and AMD

Contamination of groundwater with AMD is greatly influenced by the length of the pathway and the neutralization potential of the matrices surrounding the pathway. Unless AMD directly enters the groundwater, as can occur in flooded mine workings, contaminated water will flow through and be influenced by the vadose zone. This results in a long, slow AMD pathway which often acts to neutralize acid mine drainage.

### *Environmental Characterization*

At a minimum, pre-operational groundwater quality should be determined up-gradient and down-gradient for each groundwater stratum with the potential to be influenced by the mining operation. Standard groundwater quality, potential impact (e.g. metals, radionuclides, etc.), and hydrological parameters should be measured.

### *Preliminary Investigation/Intensive Baseline*

The intensive baseline program should involve:

- collecting water samples from each water quality station on a monthly basis for one year;
- completing an investigation to identify the instantaneous variation by collecting three to five replicates at each station for a minimum of one sampling event;
- determining the required replication for operational monitoring program by completing a power analysis on these data; and
- using the same parameter list as detailed for the environmental characterization should be used.

The data from this intensive baseline will provide the information needed to develop the operational monitoring program.

### *Operational Monitoring*

The operational monitoring program should be designed using the results of the preliminary baseline and the following procedures.

- Assess whether a stratified study design is required by analysing the preliminary survey data for evidence of seasonal patterns and/or correlations with surface water discharges.
- Determine the appropriate sampling frequency based on the degree of autocorrelation in the preliminary survey data set.

- Follow standard QA/QC procedures should be followed. The use of Quality Control samples will depend on the perceived sensitivity of the monitoring program to sampling and analytical errors or biases.
- Note that the parameter list may be reduced to the standard hydrologic characteristics, Tier I AMD indicators for water quality, and to the site-specific parameters as determined in the risk assessment modelling.

## 2.4 Section Bibliography

### 2.4 Water Quality Guidelines / Regulations

AQUAMIN 1997
BC Env. 1995
BC MELP 1993
BC MEMPR and BC MELP 1992
BC MOE 1974
CCME. 1996
Eco/Log 1992
Env. Can. 1992a

### Limnology

Becker Neitzel 1992
Mitsch and Gosselink 1993
Stumm and Morgan 1981
Wetzel 1983
Wetzel and Likens 1990

### Water Quality Study Designs and Statistics

Arnold 1970
BC AMD. 1990c
Cailas et al. 1986 Stat
Demayo and Whitlow 1986
Dunne and Leopold 1978
Hirsch 1988
Liebetrau 1979
McLeod and Hipel 1991
Pardo 1994
Skalaski and McKenzie. 1982
Smeach and Jernigan 1977
Smith 1984
Somlyody et al. 1986
Steele 1986
Valiela and Whitfield 1989

### AMD and Receiving Water Quality

BC AMD 1990c
BC Env. 1995
Bowell et al. 1995
Evaldi and Garcia 1991
Gray 1996
Kwong and Whitley 1993
Morin and Hutt 1993, 1995
Morin et al. 1995
Vinyard 1996

Ward 1986
Ward et al. 1986
Whitfield and Woods 1984
Whitfield 1988

**Analytical Considerations/Procedures**

APHA. 1992
ASTM. 1992
Hartman 1991
NOAA 1989
Plumb 1981
PSEP 1991
U.S. EPA. 1991a,b
WQNL 1985

**Water Sampling and Parameters**

Env. Can. 1983
Harsham 1995
IHD-WHO 1978
Krajca 1989

## 3.0 SEDIMENT MONITORING

For the purposes of this document, sediments are defined as the bedded particulate material, along with the pore water occupying the interstitial spaces, which has been transported and deposited in the aquatic habitat. Sediments play an integral role in the sustainment of an aquatic ecosystem. They provide food and shelter to many organisms and are essential to the mineralization and decomposition processes of potentially toxic material.

### 3.1 Trace Metal Biogeochemical Cycling

Contaminants in the sediment are locked up and slowly buried through time by sedimentation processes. Bioturbation, the movement of sediment and contaminants due to the activities of benthic organisms, and mixing due to wind energy and/or turnover events[12] (ChIV:2.1.2) slow the rate of contaminant burial. The adsorptive capacity of sediments for contaminants is dependent on the physical (e.g. particle size, mechanical disturbance), chemical (e.g., pH, redox potential, ionic composition), and biological conditions in the waterbody of interest.

---

[12] The mixing of previously thermally stratified waterbodies.

Hart (1982) provided a simplified schematic of the biogeochemical cycles of metals in lakes and rivers (Figure IV:3.1-1a,b). The following points pertain to Figures IV:3.1-1a, and 1b; the numbers refer to the numbering of stages in the biogeochemical cycle illustrated in Figure IV:3.1-1a.

Pathways:

① The biological cycle with physico-chemical sorption and active uptake of metals to phytoplankton and other particulates.

② Bacterial decomposition in the water column releases metals in a number of dissolved and particulate forms.

③ Sedimentation of particulate material. Some material may be exported via output (discharge from the waterbody) .

④ Adsorption of dissolved free metal ions ($M^{2+}$) and metal complexes or colloids (ML) onto the sediments.

⑤ Metal inputs as particulate, complexed, and free metal forms may enter directly into the biological cycle (into path 1) or,

⑥ Metals may interact directly with the sediments.

⑦ Under certain conditions sediments may act as a source of dissolved metals into the water column.

⑧ Particulate metal forms can enter the water column through perturbation of the sediments.

The cycle is similar in fluvial systems (Figure IV:3.1-1b) with some difference in the relative importance of the reservoirs and pathways. The only major difference is the greater role played by periphytic (benthic) algae which are more abundant in fluvial systems due to light penetration to the substrate.

In shallow lakes, benthic algae and macrophytes would play roles similar to stream benthic algae. The role of phytoplankton is diminished in such systems (Vanriel and Johnson 1996).

## 3.2 Biological Implications of Sediment Contaminants

The tendency of sediments to act as contaminant sinks can lead to higher sediment contaminant concentrations than in the overlying waters. This accumulation can result in biologically toxic concentrations in the sediment, which would not be directly detected in the water monitoring program. Contaminants in the sediment may persist resulting in long-term exposure to organisms living in (benthic invertebrates), or periodically contacting (benthic foraging fish species), the sediments. Contaminants in the sediment reservoir can affect organisms either through direct contact or by ingestion of contaminated particles. Exposure may occur through external contact and/or through ingestion of contaminated particles.

The measured concentration of sediment-bound contaminants is suggestive of toxic potential; however, it does not indicate the biological toxicity of the sediment. For example, sediments have been shown to exhibit differing degrees of toxicity while possessing the same total quantity of metal (Di Toro et al. 1992). To have a biological effect, sediment related contaminants must be biologically available. Contaminants are bioavailable if they are released from the sediment and are assimilated by an organism across a biological barrier. The bioavailability of a contaminant is greatly dependent on the speciation or phase distribution of the contaminant (e.g. trace metal), which for the purposes of this document will be defined as "*the distribution or partitioning of a metal among various physico-chemical forms in external medium*" (BC AMD 1993).

### 3.3 Monitoring Types

Sediment monitoring falls into three broad categories:

1) chemical characterization studies;
2) investigations of the *in situ* benthic community; and
3) sediment toxicity assays.

The latter categories are addressed in ChIV:4.0 and 6.0, respectively. This section focuses specifically on chemical characterization studies.

Sediment contaminant characterization investigations can be subdivided into four types of tests:

- total bulk contaminant analysis (ChIV:3.4);
- pore water chemistry (ChIV:3.6.1);
- trace metal phase distribution (ChIV:3.6.2); and
- elutriate tests (ChIV:3.6.3).

Of these four procedures, only total bulk contaminant investigations would be considered routine compliance/diagnostic monitoring components at most mine sites. More complex contaminant characterization investigations involve the use of pore water investigation, and fraction/extraction procedures, where the objective is not only to quantify contaminant concentrations in the sediment, but also to assess the *in situ* bioavailability of these contaminants. These techniques require specialized equipment and personnel and are therefore classified as investigative monitoring procedures. However, recent scientific advances may shift the monitoring emphasis of sediments towards the measurement of bioavailability of contaminants (SETAC - December 1996). Elutriate tests, are and have recognized standard protocols. However, they are generally only of value in assessing sediments prior to disposal or dredging operations.

### 3.4 Total Bulk Contaminant Characterization

**Total bulk contaminant chemical characterization of sediments is recommended as a Tier I AMD monitoring component.** The collection and chemical analysis of sediment samples has long been employed by the mining industry for both exploration and monitoring purposes. Lake sediment surveys have been used as a preliminary exploration tool to delineate regional zones or belts of increased metal content, which may be correlated with features of economic interest (e.g., Hornbrook and Garrett 1976; Sopuck et al. 1980; Coker and Dunn 1983). Sediment contaminant characterization studies, in one form or another, have also served as regular core components of most mine monitoring programs. This often involves monitoring total contaminant concentrations or loadings in the sediment at reference station(s) and at potential contaminant receiving stations. They are of greatest value as a component of contaminant-loading monitoring designs which incorporate water-quality monitoring as detailed in ChIV:2.1.4.2.

## 3.4.1 Environmental Characterization or Preliminary Survey

### 3.4.1.1 Preliminary Mapping

During the environmental characterization phase a comprehensive aquatic habitat classification mapping program should be completed for potential reference and influenced waterbodies (TSN 4.3). The required information is listed in Table IV:3.4-1 with the main sediment related requirements addressed in further detail within this subsection.

***Lakes***

- Develop bathymetric maps of the potential impact and reference waterbodies in the greater study area.
- Identify sediment types and their distribution in these waterbodies with special emphasis on locating areas of fine sediment deposition.

Preliminary sediment mapping may be completed using acoustic surveys or through a low intensity sediment sampling program. **Acoustic surveys** (e.g., echo sounding, ground penetrating radar, seismic reflections or refractions) may be completed to characterize the surficial sediment layer (e.g., sand, gravel, silt) and the subsurface sediments. These techniques require special equipment and may need to be interpreted by a geophysicist.

A **low intensity sediment sampling** program involves collecting samples along a series of transects (located using the bathymetric data) to identify areas of fine-grained sediment deposition (usually in the deepest regions). These regions have the greatest potential for long-term contaminant accumulation.

Selection of the number of transects, and of the sampling intensity along a transect, can be a subjective process. At a minimum, sampling stations should be located along a transect situated on the long axis of the lake. Stations should then be sampled on a few transects located perpendicular to the long axis. Stations should be positioned on these transects so that a wide range of depth zones is represented. This means more complex bottom morphometry requires a denser network of stations (Mudroch and Azcue 1995). Lakes with irregular shorelines (no clear central axis) or where the study area is restricted to a river mouth or point source may be best served by positioning transects in a ray-like pattern. Examples of possible transect positioning are shown in Figure IV:3.4-1.

A sampling algorithm has been proposed (Rowan et al. 1995) for determining the general positioning and the number of stations required to optimize sampling effort in lake sediment surveys. This algorithm requires only bathymetric and catchment parameters. Rowan et al. (1995) should be consulted for further information.

*Rivers*

- Complete a physical investigation of any rivers or streams (e.g. foot or aerial reconnaissance) to subdivide the river into depositional and erosional zones.
- Identify the sediment types in the depositional and erosional zones.

Detailed surface sediment maps categorizing the distribution of sediment types (e.g. organics, silts, sand, gravel, etc.) are required if the mining activities are expected to result in significant changes in sedimentation patterns. Changes in sedimentation patterns may result, through alterations of flow patterns, or where large releases of sediments are expected (e.g., placer operations).

### 3.4.1.2 Analytes or Parameters of Interest

At a minimum, the samples collected in the preliminary survey should have their physical characteristics determined. This involves recording the following information:

- **Physical logging:**
  - Equipment used for core collection.
  - Name of the operator who collected, handled, and split the core.
  - Length of the retained core.
  - Description of splitting methods.
  - Thickness of sediment units (may be based on colour).
  - Consistency (e.g., soupy, soft, medium-firm, etc.).
  - Texture (e.g., silty sand, sandy clay, etc.).
  - Structure (e.g., graded bedding, cross bedding, etc.).
  - Presence and description of organic matter.
  - Odour.
  - Appearance of oil, coal dust, ash, etc.
  - Presence of carbonates (see Mudroch and Azcue 1995).

Recommended additional parameters.

- **Sediment grain size**
  - important in determining the sorptive capacity (e.g for heavy metals) of sediments.
  - used to indicate the potential availability of contaminants to benthic organisms through ingestion of fine-grained materials.
- **Total Organic Carbon (TOC) or % loss on ignition (%LOI)**
  - potential metal-adsorption capacity of the organic load in the sediment.
- **Wet Weight/Dry Weight**
  - for estimating the wet weight that will meet dry weight analytical requirements.

Chemical characteristics would also be of value; however, if the budget is limited analysis of chemical constituents may be delayed until the intensive baseline programis implemented.

### 3.4.2 Development of Operational Monitoring Program

#### 3.4.2.1 Station Locations

The specific objective of the monitoring program is of key importance when determining sampling strategies. Three such objectives are outlined as follows:

**Objective A**: To monitor the spatial and temporal influence along a drainage of a point source release (most common AMD concern).
Compliance /Diagnostic monitoring.

**Objective B**: To quantify contaminant loadings and determine contaminant budgets for a lake (see ChIV:2.1.4.2 for water quality concerns).
Diagnostic/Investigative monitoring.

**Objective C**: To identify or locate sites of contaminated sediments on a quantitative spatial and/or temporal basis. Identification of contaminant "hot spots."
Diagnostic/Investigative monitoring.

To meet **Objective A** the station distribution pattern should be based on the influence of factors affecting contaminant dispersion (e.g. currents) (Env. Can. 1994). Hence, plume delineation or

**plume** modelling (see ChIV:1.1.2.3) results should be consulted when locating sediment stations. The key points are listed below.

- Locate stations in fine-grained depositional zones identified in the preliminary survey.
- Select stations with sediments of similar physical composition/dynamics.
- Locate the first impact station at the nearest depositional zone to the point of release to the receiving environment.
- Locate additional stations within the modelled mixing zone at increasing fixed intervals downstream.
- Locate the last downstream station should be located in a depositional site considered to be beyond the point of potential environmental effects.
- Locate reference station(s) at depositional sites identified in the extensive survey as being similar to the impact stations. Since sediment chemical composition of lakes can vary dramatically in response to local mineralization (hence its use as an exploration tool), the study design should incorporate multiple control or reference sites/waterbodies.

For example, point source releases into rivers are best monitored by locating stations in the **nearest** deposition zones (e.g., pools or small riverine lakes) approximating fixed distances (Mudroch and MacKnight 1991) or modelled water quality dilutions, following a geometric progression (i.e., X, 2X, 4X, 8X, . . .). If the point of release is into a large waterbody where dispersion may not be unidirectional (e.g., Figure IV:1.1.1), radiating patterns should be employed (e.g. Figure IV:3.4-2) with stations located at selected intervals along the radiating axes. If plume delineation has been completed, additional transects may be concentrated in the predicted path of the plume.

**Objective B** involves quantifying the compartmental (water, sediment, biota, etc.) distribution of the specific contaminant(s) of concern. The sediment component of such an investigation requires a stratified sampling design. Sediment mapping completed during the preliminary survey is used to select a number of sediment sub-units or stratum with sampling stations divided between these strata. The most common factors used to subdivide strata are those associated with contaminant accumulation, such as dominant particle sizes and organic composition.

**Objective C** needs are best met using a systematic or regular grid study design. This involves dividing the study area into a sampling grid (squares, rectangles, or triangles) and collecting within the grids. Gilbert (1987) discusses the appropriate sampling design for identifying suspected "hot spots" where the following three questions are to be addressed:

1) What grid spacing is needed to hit a suspected hot spot with a specified confidence?
2) For a given grid spacing, what is the probability of hitting a hot spot of a specified size?
3) What is the probability that a hot spot exists when no hot spots were found by sampling on a grid?

***Accurate Relocation***

The ability to relocate a sediment sampling station accurately is required for any sediment- monitoring program interested in temporal changes in contaminant concentrations. Information on the available options for siting in a station location for future sampling events is provided in TSN 9.0.

### 3.4.2.2 Sample Replication Requirements

Once the stations (Objectives A and C above) or strata (Objective B above) have been identified the number of replicate samples required within each station or strata should be determined.

**Objective A** monitoring programs are designed to test the null hypothesis ($H_o$) that there is no difference between the mean sediment contaminant concentration, either spatially (amongst the station(s)) or temporally (between sampling periods). Such monitoring programs require the calculation of the mean contaminant concentration(s) within a station and an error estimate for the mean, as it is not possible to test for differences among stations if within- station variability is not accounted for. Therefore, once the location of a sampling station has been determined, the number of sample replicates to be collected **within** that location must be decided.

**Objective B** monitoring programs require replication within a stratum (# of stations) to identify the confidence which may be placed in the estimate of total contaminant loadings for a specified stratum and to support any spatial (between strata) or temporal (between study periods) statistical comparisons.

The number of stations placed within a stratum may be determined in one of two ways. The simpler involves distributing the stations amongst the strata based on the percentage contribution of that stratum to the lake as a whole (e.g., 40% silt: 4 of ten budgeted stations randomly located in silt substrates, Figure IV:3.4-3). The statistical approach involves completing of a preliminary study designed to measure the variance within each of the defined strata. Power analysis is then completed to determine the number of samples required **per stratum** to achieve an estimate of the mean contaminant concentration for each stratum within a specified confidence limit (Gilbert 1987).

### *Considerations for Sample Replication*

Complete statistical power analyses on preliminary sampling data to identify the number of replicates required to meet the predetermined level of precision (Green 1979; Mudroch and Azcue 1995). If statistical determination of the number of replicates identifies sampling intensities beyond the technical and financial capabilities of the program, use one or both of the following alternatives:

- Determine whether additional replicates may be accommodated in the program by refining the parameter list and/or decreasing the number of sampling stations.

- Re-assess the level of precision and confidence required by the monitoring program based on the toxicity of the contaminant(s) of concern. If a large increase in concentration is required to approach levels of concern, lower confidence limits may be acceptable. If minor increases could have marked environmental implications then high precision and confidence is required.

Ensure that the minimum number of replicates collected at a station be three to five, with error estimates provided for each calculated mean. The literature listed in the section bibliography should be consulted for additional information on statistical determination of within-station/strata replication for monitoring programs.

### 3.4.2.3 Particle Size Considerations

There are a number of physical attributes of interest when evaluating sediment chemical data, such as grain size, surface area, specific gravity, surface charge, bulk density, sheer stress, porosity, and permeability. The most important for interpreting sediment chemical data is grain or particle size, as it tends to integrate all of the other factors (Horowitz 1991a, Mudroch and Azcue 1995). Table IV:3.4-2 shows the commonly used sediment particle size classes and the analytical options for determining particle size.

Sediments in depositional zones usually consist of sand or finer materials (<2,000 • m). Silt and clay particles (approx. <63 • m) are of particular importance in determining a sediments' capacity for concentrating and retaining contaminants. Compared to coarser particles, these fine-grained particles have a greater affinity for inorganic contaminants due to both physical (e.g., relatively large surface area) and chemical (e.g., geochemical substrates) factors. Their importance in sediment monitoring is further enhanced by their greater direct association with the benthic macroinvertebrate community through ingestion. Hence, bulk contaminant monitoring programs should be concentrated in depositional zones dominated by fine particulate material, which for the purposes of this document are defined as particles ≤63 • m.

Direct comparisons of contaminant concentrations between sampling sites is not possible unless sediment particle composition is consistent. The presence of coarser materials (> 63 • m) serves to dilute the total contaminant concentrations, especially for metals and trace metals which have a high affinity for the fine particles (Mudroch and Azcue 1995). Three procedures have been proposed to address this concern:

1) chemical analysis of separate particle size classes;
2) correction using particle size distributions in a bulk sample; and
3) correction using the concentration of a conservative element in the bulk sample.

Of these three approaches, chemical analysis after physically separating the sample into fine and coarse fractions provides the most accurate results for metal concentrations. Corrections based on particle size and conservative elements tend to over- or under estimate metal concentrations and often require the collection of a large number of samples to confirm the power of the correction. A brief discussion on these three options is provided in the following paragraphs. Final decisions addressing particle size concerns for a routine compliance or diagnostic monitoring program should be arrived at in consultation with the relevant regulatory body to identify site-specific protocols.

### *Chemical Analysis of Separate Size Classes*

Samples collected from different stations may be separated into defined size classes (Table IV:3.4-2a) with chemical analyses completed on each of the size classes. When separating the fine fraction into classes <63 • m it can be difficult to collect sufficient material to meet analytical requirements. Fine size fractions should only be compared when they have been separated using the same technique and pre-treatment processes (Table IV:3.4-2b). For example, Figure IV:3.4-4 shows the markedly different size distributions which resulted when air elutriation techniques were compared to chemically dispersed pipette analyses (Horowitz and Elrick 1986). Hence, an experienced sedimentologist should be consulted prior to incorporating subdivision of the fine fraction into a study program. For routine monitoring purposes, analyses of the fine fraction (≤63 • m) as a whole is adequate (Mudroch and Azcue 1995). The organic component (loss on ignition) must be determined prior to particle size analysis if the separation technique requires ashed samples.

*Correction Using Particle Size Distribution*

Instead of analyzing physically separated particle sizes, the contaminant concentrations determined for a bulk sediment sample may be corrected based on the particle size distribution in the sample. This approach assumes that the relationship between the contaminants of interest and the fine particle component of the sediment is linear. The contaminant concentration is then plotted against the percentage contribution of the fine particles (≤63 • m) to the whole sample. Should a linear regression of this relationship exhibit a high correlation coefficient and a *p* value[13] ≤0.05 or 0.01, then normalization based on the fine particle component may be appropriate (Mudroch and Azcue 1995). Unfortunately, this approach requires a large number of replicates (10 to 15) from a study area and a suitable regression is difficult to achieve if particle size ranges in the study area are limited. The coarse component may exhibit a high correlation with the organic composition.

*Correction Using a Conservative Element*

In this approach the contaminant concentration is corrected using the concentration of a conservative trace element in the bulk sediment sample. The conservative element used for normalization should be a significant constituent of one or more of the major fine grained carriers of the contaminant and must reflect its variation with different sediment particle sizes. Aluminum has been used as a fine particle proxy, as it occurs naturally in alumino-silicates which are mainly found in the silt-clay fraction (Mudroch and Azcue 1995). Cesium has also been presented as an appropriate proxy for correcting heavy metal concentrations to the ≤20 • m fraction due to its low concentration in quartz (the main constituent of sand fractions) relative to its concentration in clay materials (Ackermann 1980). Prior to using a conservative element for normalizing contaminant concentrations the dependence of the proposed proxy to the fine fraction must be determined.

### 3.4.2.4 Sample Depth Profile

---

[13] P value is the probability of incorrectly determining that the regression slope is significantly different from zero, i.e., a Type I error.

A number of biotic and abiotic factors such as bioturbation, sedimentation rates, and turn-over events, influence the vertical accumulation of contaminants. The upper 0 - 5 cm layer of sediment is the sampling horizon of interest for most sediment monitoring programs, where historical contaminant accumulation is not a significant priority (Env. Can. 1994). The upper 2 cm of this horizon will consist of the most recently accumulated contaminants and represents the region with the greatest benthic macroinvertebrate biomass and activity (Env. Can. 1994). There are currently no standard regulations stating the depth of the vertical horizon which should be analyzed for monitoring purposes. The sampling protocols have tended to vary among different jurisdictions and among mine sites within the same jurisdictions.

Currently there are ongoing discussions between regulatory agencies and industry (SERM, AECB, DFO, AETE, etc.) addressing concerns over the appropriate depth zones for sediment core fractions. A number of protocols have been proposed ranging from 1 cm slices for the top 15 cm to analysis of a sample consisting of a full Ekman Dredge. Until standardized regulations are decided upon it will be necessary for each mine operation to consult with the relevant regulatory body to identify site-specific[14] protocols.

### 3.4.2.5 Where and When to Sample

Sediment monitoring in areas of fine particle deposition serves to provide information on the long-term accumulation of contaminants. This can be achieved using a fixed sampling frequency ranging from every three to five years, depending on site-specific concerns.

Compared to water-quality monitoring, sediment monitoring in long-term depositional sites is less sensitive to short-term environmental events (e.g., source water-quality hysteresis effect (ChII:1.5); however, certain seasonal factors must be accounted into the sediment program. Periods of instability at the sediment-water interface such as seasonal turn-over or mixing events and periods of high flow or discharge in rivers should be avoided.

### 3.4.2.6 Sampling Platform

There are essentially three options for sampling platforms, the ice surface, a boat or in the water (a diver). Sampling from the ice has the benefit of providing a stable platform and a non-restrictive

---

[14] For example, in lakes with a very flocculent sediment-water interface, it may not be possible to accurately sample the upper 1 cm.

working area. If small samplers are used a single hole augered through the ice is often sufficient. If larger holes are required overlapping holes can be drilled in a round or square pattern from which the central ice plug can be removed using ice tongs. Winch or pulley frames can be set up on the ice surface if required.

The smaller grab and core samplers can be operated from a boat. Depending on the size of the boat and the water depth, portable winches, hand reels, or line keepers can be used. Operating from a boat makes establishing a well defined sample grid pattern difficult (i.e., wind-induced boat movement). Often it is more efficient to sample along a linear pattern established by the boat anchor line, taking care not to sample too close to the disturbed anchor site.

The third option is to sample using a diver. Sediment sampling investigations focussing on the sediment water interface require undisturbed samples. The ability of a diver to observe and control the placement (visually select site, detect and reject a poor sample, etc.) of the sampler and to make notes and photo/videotape the site can greatly improve sampling efficiency/accuracy. The value of using a diver declines under conditions of low visibility, very shallow or deep water, or the presence of currents. Care must be taken to prevent the divers' movement from disturbing the surficial sediments.

### 3.4.2.7 Selection of Sampling Equipment

There are two types of commercially available sediment samplers:

- grab samplers (Figure IV:3.4-5)
  - to collect surface sediments for the determination of horizontal distribution; and
- core samplers (Figures IV:3.4-6 and 3.4-7)
  - determining horizontal and vertical distribution of parameters.

No single sampler is suited for all possible situations. In terms of sample integrity (minimal disturbance), generally grab samplers are considered less efficient than corers (ASTM 1991). TSN 4.5 addresses the most common commercially available sediment samplers with Tables IV:3.4-3 and 3.4-4 providing summaries of their advantages and disadvantages and recommendations for specific samplers. Figure IV:3.4-8 provides a simplified flowchart for selecting the most appropriate sampler. For an in-depth review of sampler types the reader is referred to Env. Can. (1994), and Mudroch and Azcue (1995).

Grab samplers are suitable for the preliminary mapping of the surficial sediments and may be used for gross bulk chemical analysis. Core sampling is recommended when the objective is to characterize the chemical composition of specific vertical horizons (e.g., top 2 cm). Vertical horizons are sometimes collected using grab samples, which are then subsampled using small core tubes, when brought to the surface. This methodology is not recommended due to the greater potential for washout of the surficial sediments, when using grab samplers compared to core samplers. Loss of the fine surficial sediments can also result when using core samplers, if care is not taken to minimize the shock wave when delivering the core to the sediment.

### 3.4.2.8 Sample Wet Weight Requirements

Properly estimating the required wet field weight for analysis is critical, especially when returning to the site is costly. The dry mass requirements for a number of parameters are provided in Table IV:3.4-5.

If the sampling program does not require distinct vertical horizons, then a single bulk grab sample usually provides sufficient material to meet full analytical requirements. Sampling of defined vertical horizons is often hampered by small sample volumes and high water content and will often require compositing of a number of core horizons. The number of separate collections required to meet a single sample's mass requirements is dependent on the thickness of the desired sampling horizon (vertical sectioning or whole sample), diameter of the core or length and width of the grab sampler, the sediment water content, and whether the sample will be subdivided into specific particle sizes. Table IV:3.4-6 from Mudroch and Azcue (1995) provides estimates of the dry mass that would be collected from the upper 1 cm of various sized core tubes assuming 90 - 95% water content. Site-specific information on water content should be available from the preliminary survey (ChIV:3.3.1.1: Preliminary survey parameters).

### 3.4.2.9 Analytes or Parameters of Interest

The analytes or parameters of interest will often be site- and objective-specific. Hence, only generic parameters will be discussed in this section, with a more comprehensive list of potential parameters provided in Table IV:3.4-5. The parameter list will also differ somewhat depending on whether the mine is pre-operational, operational or in the decommissioning phase.

***Intensive Baseline (pre-operational phase)***

The purpose of the intensive baseline investigation is to collect detailed, replicated data on the physico-chemical characteristics of the sediment quality at fixed points in time (pre-impact, specific sampling season) and place (the monitoring stations) against which future surveys (operational monitoring program) will be compared.

Minimum requirements:

- Match the protocols of the proposed **operational monitoring program**.
  - i.e. sampling techniques, sampling season, parameters, etc.

As it is not possible to collect baseline data once mining operations have commenced, it is prudent to collect data on any additional parameters not at present incorporated in the operational monitoring program which may be useful in the future.

Recommended Sediment Analytes for the Intensive Monitoring Program are:

- **Physical logging**
- **Particle size distribution**
- **Total Organic Carbon (TOC) or % Loss On Ignition (%LOI)**
- **Wet weight/Dry weight**
- **Sulphides**
  - increase in sediments as a result of rapid erosion (natural or anthropogenic) and industrial inputs.
  - potentially toxic, however, also have the potential for binding trace metals making them unavailable for biological uptake (see ChIV:3.6.2.2).
- **Ammonia ($NH_3$)**
  - natural product of organic decomposition.
  - present as $NH_3$ (toxic state) or $NH_4^+$, depending on pH.
  - in acidic conditions $NH_4^+$ can begin to accumulate in the sediments.
  - a potentially toxic release of $NH_3$ can result when acid drainage inputs cease, and pH shifts towards neutrality.
- **Metal concentrations**

  Metals are the primary contaminants of concern in the sediments for acid mine drainages. Determine baseline concentrations for the following metals:

  - metals in the Metal Mining Liquid Effluent Regulations (MMLER) (arsenic, copper, lead, nickel, and zinc).
  - metals of site-specific concern (e.g., metals identified as possible components of acid drainage).
  - metals associated with the release of treated mill effluent (e.g., barium used in the treatment of uranium mill effluent).
- **Site-specific parameters**

  Examples:

  - uranium, radium $^{226}$ , lead$^{210}$ and polonium$^{210}$ for the uranium mining industry.

  - cyanide (CN$^{-}$) for gold cyanide heap leach facilities.

Additional parameters requiring further technical expertise and equipment.

- **pH and Eh (Redox State)**

  - pH and Eh should be measured in the field immediately upon sediment retrieval. -important to the geochemical processes occurring in the sediment.

  - generally adsorption of cationic metals increases with an increase in pH, whereas adsorption of metals present as anions increases as pH is reduced.

  - Eh measurements are of particular importance if information on the state of metal speciation and sediment oxidation is required.

Note: the representivity, accuracy and precision of pH and Eh measurements are questionable due to sampling disturbance and the reliability and sensitivity of the meters and probes (Mudroch and Azcue 1995). It is recommended that the literature listed in the section bibliography be consulted prior to deciding to incorporate these two parameters in routine monitoring programs.

### *Operational Monitoring Program*

The objective of the operational phase of the sediment monitoring program is to detect accumulation or loadings of contaminants in the sediments. The key decision is to determine the total concentration(s) which the system can sustain before ecological effects occur. These predetermined[15] concentration(s) serve as a "trigger" leading to the initiation of Tier II or III contaminant bioavailability investigations.

---

[15] Should be a **conservative** estimate of sustainable concentration(s).

The federal government is scheduled to release sediment quality guidelines (SQG) which will consist of calculated Threshold Effects Levels (TEL) and Probable Effect Levels (PEL). A site-specific sediment contaminant concentration trigger bounded by the TEL and the PEL could be agreed upon between the site proponent and the regulatory agencies (Figure IV:3.4-9). Exceeding of this trigger point should lead to initiation of additional investigative monitoring such as sediment toxicity assays (Tier II or secondary components) and contaminant bioavailability investigations.

Suggested Sediment Analytes for the Operational Monitoring Program are:

- **Physical logging**
- **Particle size distribution**
- **Total Organic Carbon (TOC) or % Loss On Ignition (%LOI)**
- **Wet weight/Dry weight**
- **Sulphides**
- **Ammonia**
  - monitoring if acid releases commence.
  - increases in importance after successful remedial action has been taken (see parameters for intensive baseline).
- **Metal concentrations**
  - metals designated for sediment monitoring in the operating permits.
  - site-specific metals identified as possible components of AMD
  - consider tiering if a significant release of multiple metals is expected in an acid event.
- **Site-specific parameters**
  As detailed for intensive baseline.

### *Decommissioning*

During the decommissioning phase, sediments monitoring should be completed to determine whether the sediments are continuing to act as a sink and burying the contaminants; or whether the flow has reversed, with the sediments acting as the contaminant source for the water. Sediment sampling, in conjunction with a water-quality program, can be used to monitor contaminant loadings and identify any flow reversal of contaminants.

Suggested Sediment Analytes for the Decommissioning Monitoring Program are:

- **Physical logging**
- **Particle size distribution**
- **Total Organic Carbon (TOC) or % Loss On Ignition (% LOI)**
- **Wet weight/Dry weight**
- **Sulphides**
- **Ammonia**
  - if acid generation has occurred.
- **Metal concentrations**
  - those identified by the operating program as present in ecologically significant high concentrations in sediments (Defined in consultation with the regulators).
- **Site-specific parameters**
  - parameters of concern identified by the decommissioning risk assessment.

### 3.5 Quality Assurance and Quality Control

Containers and other equipment used for processing samples can be significant sources of contaminants. Table IV:3.4-7 provides a list of sample containers and preservation procedures for a number of the most commonly measured sediment parameters. Samples requiring shipment to a laboratory should be shipped at approximately 4°C[16] (e.g. coolers with ice packs). Additional documentation which should be provided with the field notes is shown in Table IV:2.1-3. The standard QA/QC procedures outlined in ChII:2..6 should also be followed where applicable .

Special attention is required to prevent sample contamination when:

- handling the sample in the field (e.g. preservation, sectioning, measuring pH, etc.);
- the parameters of interest are present in relatively low concentrations (e.g., cadmium, mercury, and many organic contaminants);
- there is the potential for cross-contamination (e.g., when samples from contaminated and uncontaminated sites are collected during the same field trip); and
- the parameter of interest is highly volatile (e.g. Hg and some organic contaminants).

---

[16] Exceptions to this guideline are for mercury and organic contaminant tests for which sediments must be frozen to at least -20°C.

To minimize contamination due to handling, standard practices should be followed:

- clean and decontaminate sampling devices between sites;
- rinse the sampling equipment in the ambient water at the sampling site; and
- carefully package samples stored or shipped together to prevent cross contamination.

Sample containers and core tubes require special cleaning procedures. Qualified laboratory specialists should be consulted to confirm that the decontamination procedures are appropriate for the contaminants of interest.

## 3.6 Investigative Monitoring Techniques

The following information is provided to introduce mine personnel and managers to investigative techniques. This will provide a starting point to open discussions with contractors and regulators should these techniques be required or proposed as possible approaches to specific sediment contamination concerns.

### 3.6.1 Pore Water Chemistry

Sediment pore water or interstitial water is defined as the water filling the space between sediment particles and not held by surface forces to sediment particles. Pore water acts as the link between sediments and the overlying water and is one of the primary contaminant exposure routes for benthic organisms. Sediment pore water chemistry is of interest in subaqueous tailings investigations (ChIII:2.3.3.1) related to source pathways and investigations of contaminant bioavailability in the receiving environment.

Analysis of sediment pore water can be used to obtain information on:

- chemical changes occurring in the sediment,
  - equilibrium reactions between the sediments solid phase and pore water
  - fluxes of contaminants into the sediment/water interface and overlying water; and
- bioavailability of nutrients and contaminants.

Many indirect and direct collection methods exist; however, until standard methods have been determined, pore water investigations should be considered investigative components exercised on a site-specific basis. For a short review and discussion on pore water sampling, the reader is referred to Mudroch and Azcue (1995).

With the exception of the sediment/water interface, sediments are generally anoxic and tend to be rapidly oxidized upon exposure to the air. This oxidation tends to alter the chemical speciation of redox-sensitive components of the pore water, with many dissolved elements precipitating. Hence, handling must be completed in an oxygen-free environment and analysis must be completed rapidly after sampling. There are four standard techniques for pore water investigations as shown in Table IV:3.6-1. A brief discussion will be provided for each in the following paragraphs.

### 3.6.1.1 Indirect Method

considered quantitative and evaporation is a significant problem due to the slow rate of pore water removal. Extraction by squeezing is only suitable for the collection of representative pore water samples rather than quantitative samples (Mudroch and Azcue 1995).

#### 3.6.1.2 Direct Methods

The main handling problem associated with the indirect pore water approaches has been identified as sediment oxidation. To minimize this sampling artifact, *in situ* pore water sampling procedures have been developed. These samplers employ two basic approaches, suction and dialysis.

**Suction** samplers generally function by placing a probe with sampling ports into the sediment. Pore water is drawn through the Teflon or ceramic filters of the sampling ports using vacuum pressure generated by syringes, the release of pressurized gases, or vacuum pumps. The primary difficulty with these samplers is determining the appropriate pore size for the sampling port filter. Clogging becomes a problem if the pore size is too small, while contamination with fine-grained sediment particles is a problem if the pore size is too large.

**Dialysis** samplers are based on diffusion-controlled transport, where equilibration occurs across a dialysis membrane separating the pore water and the water in the sampler. The most common form of dialyser is the Peeper developed by Hesslein (1976). The basic design of a peeper is shown in Figure IV:3.6-1 and usually consists of two acrylic sheets: one 1.3 cm thick main body and a 0.3 cm thick cover. Elongated cells are machined into the main body usually at approximately 1 cm depth intervals. A dialysis membrane is sandwiched between the main body and the cover sheet. The cover sheet has slots cut in it which align with the cells in the main body. The cells in the main body are filled with deoxygenated double distilled water. All handling of the peeper prior to placement is under oxygen-purged conditions.

The Peeper is driven into the sediments and the water in the cells is allowed to equilibrate in place for a designated time period (usually ranging from 6 to 30 days). Upon retrieval, the cells are sampled using a syringe. Collection from the cells should be completed within 5 minutes of removal from the sediments.

### 3.6.2 Trace Metal Phase Distribution

#### 3.6.2.1 Sequential Extractions

While total metal concentration may be determined in bulk contaminant analyses, the proportional distribution of the metals among the various phases will differ under various environmental conditions. The toxicity, or bioavailability of the metals is primarily determined by the chemical partitioning and phase distribution. In order of increasing bond strength, they may be:

1) adsorbed (e.g., to clays, humic acids, etc.);
2) bound to carbonates;
3) bound to iron and/or manganese oxyhydroxides;
4) bound to organic matter;
5) bound to sulphide; or
6) matrix bound (bound in lattice positions).

Sampling procedures and laboratory analyses for sequential extractions are relatively complex, requiring sampling and handling under anoxic conditions (e.g,. nitrogen purged environments). The basic protocol consists of leaching the sediments with progressively stronger reagents to determine the metal concentrations in each of the extracts. The mildest extraction is completed first, with the solids recovered by filtration, washed and used for the next extraction. This is repeated until the entire predetermined series of extractions has been completed. The data are then used as a correlate for the bioavailability of the metals of interest.

### 3.6.2.2 Acid Volatile Sulphides (AVS)

The bioavailabity of metals in sediments is related to the chemical activity of the metal in the sediment-interstitial water system. Recent research suggests that biological effects can be correlated to the divalent metal activity (from Di Toro et. al. 1992) and that under anoxic conditions this activity in the sediment-interstitial water system is strongly influenced by the sulphide concentrations in the sediment. The sulphides most responsible for this control are termed acid volatile sulphides (AVS) due to their ability to be extracted with cold hydrochloric acid. The fraction of metals extracted along with the AVS are referred to as Simultaneously Extracted Metals or SEM. It has been shown for a number of metals that when AVS concentrations exceed those of metals simultaneously extracted with the AVS, metal concentrations in the interstitial water are low, and toxicity is not observed (Di Toro et al. 1992, Ankley et al. 1996).

Experiments have demonstrated that when the molar concentration of SEM to AVS was less than one, that no acute toxicity occurred (i.e., [SEM]/[AVS] < 1); however, when [SEM]/[AVS] >1, then the mortality of sensitive species increased (Di Toro et al. 1992). Currently, this method may be used to predict the **lack** of bioavailability for cadmium, copper, nickel, lead, and zinc (if [SEM]/[AVS] < 1). However, it cannot be used to predict bioavailablity (still may not be bioavailable when [SEM]/[AVS] > 1) as other factors such as organic carbon, and manganese oxides will also influence bioavailability .

Recently, recommendations have been made to the Science Advisory Board of the U.S. EPA that interstitial water chemistry (porewater) and /or metal-AVS relationships be utilized to assist in the derivation of Sediment Quality Guidelines (Ankley et al. 1996).

Key Points on the Potential of SEM/AVS for Monitoring:

- proposed that when [SEM]/[AVS] <1 metals are **NOT** biologically available.
- proposed that when [SEM]/[AVS] >1 metals **MAY** be bioavailable.
  - e.g. an area with low [SEM] and no [AVS] does not necessarily indicate a toxic sediment.
- standardized methods are lacking (see Brumbaugh and Arms 1996 newly proposed QC).
- concept based on assumption of anoxic conditions which may not hold at the micro-habitat level (BC AMD 1993).
- further research required, but it is rapidly developing as a promising monitoring tool.

### 3.6.3 Elutriate Tests

Elutriate tests were initially developed to assess the effects of dredging operations on water quality, but may also be applied to situations where the re-suspension of sediment-bound toxicants is a concern (Env. Can. 1994). The tests are used to provide information on the **potential** effects of a disposal operation on water quality (BC AMD 1990) and are not considered appropriate for determining pore water or bedded sediment contaminant concentrations or toxicity (Env. Can. 1994).

Elutriate tests are relatively simple short-term sediment leaching tests. A known volume of sediment (collected via a grab or corer) is combined and agitated with a known volume of site water in an approximate volumetric ratio of 4:1 (e.g., 1000 g dry sediment: 4000 g). This mixture is agitated for 30 minutes and then allowed to settle. The supernatant is decanted and centrifuged. The elutriate (supernatant) is now available for chemical analysis (with or without filtration depending on the end-use) or for use in biological assays. Elutriate bioassays have been found to be **more**, **similar**, or **less** toxic than pore water based on differences in bioavailability between the two test types (Env. Can. 1994).

## 3.7 Section Summary

Sediments often act as contaminant sinks, and typically contain higher concentrations of contaminants relative to the overlying water. This can result in the accumulation of contaminants in the sediments to biologically toxic levels, which would not be directly detected in a water-quality monitoring program. These sediment bound contaminants may persist resulting in long-term exposure to organisms living or feeding in, or periodically contacting the sediments.

### Total Bulk Contaminant Characterization

Total bulk contaminant chemical characterization of sediments is recommended as a Tier I AMD monitoring component. Bulk sediment characterization is best completed in conjunction with water-quality contaminant-loading monitoring programs.

#### *Environmental Characterization / Preliminary Mapping*

The preliminary survey should involve sediment mapping of the greater study area (control/reference waterbodies) and a review of the historical information for the study area. At a minimum, samples collected for mapping purposes should receive physical logging. Additional parameters that should be considered are particle size determination, analysis for total organic carbon (or loss on ignition) and moisture content.

#### *Operational Monitoring Program*

The specific objective of the monitoring program is of key importance when determining sampling strategies. Three such objectives are outlined below.

**Objective A**: To monitor the spatial and temporal influence along a drainage of a point source release (most common AMD concern).

- Compliance / Diagnostic monitoring.

**Objective B**: To quantify contaminant loadings and determine contaminant budgets for a lake.

- Diagnostic / Investigative monitoring.

**Objective C**: To identify or locate sites of contaminated sediments on a quantitative spatial and/or temporal basis. Identification of contaminant "hot spots."

- Diagnostic / Investigative monitoring.

***Locating Stations***

**Objective A** requires a station distribution pattern that reflects the influence of factors affecting contaminant dispersion (e.g., currents). Hence, plume delineation or plume modelling results should be consulted when locating sediment stations.

Stations should be located in fine-grained depositional zones with the stations possessing sediments of similar physical composition/dynamics. The first impact station should be located in the nearest depositional zone to the point of release to the receiving environment. Additional stations should be established at increasing intervals downstream, with the last downstream station located in a depositional site considered to be beyond the point of potential environmental effects.

Reference station(s) should be located at depositional sites identified in the extensive survey as being similar to the impact stations. Multiple control or reference sites/waterbodies are preferred.

**Objective B** involves quantifying the compartmental (water, sediment, biota, etc.) distribution of the specific contaminant(s) of concern. Sediment mapping completed during the preliminary survey is used to select a number of sediment sub-units or strata with sampling stations divided between these strata.

**Objective C** requires a systematic or regular grid-sampling study design. The study area should be divided into a sampling grid (squares, rectangles, or triangles) with collections made within the grids.

***Additional Considerations***

The required number of **sample replicates** per station or strata is identified by completing power analysis on the preliminary sampling data. Direct comparisons of contaminant concentrations between sampling sites are not possible unless **sediment particle composition** is consistent or corrected for by either:

- chemical analysis of separate particle size classes;
- correction using particle size distributions in the bulk samples; and/or
- correction using the concentration of a conservative element in the bulk samples.

Until standardized regulations are developed identifying the specific **depth horizon** to be sampled it will be necessary for each mine operation to consult with the relevant regulatory body for site-specific protocols. The collection of the upper two cm horizon is proposed as a suitable compromise between the cost and difficulties associated with shallow sampling horizons (e.g., 1 cm fractions) versus the dilution effect of including sediments likely deposited prior to the commencement of operational contaminant releases.

On-site characteristics and the objective of the sediment program will determine whether the samples should be collected from the **ice surface**, **boat**, or by a **diver**, though the use of a diver to visually assess the *in-situ* sediments is always beneficial. **Core** sampling is recommended when the objective is to characterize the chemical composition of specific vertical horizons (e.g., top 2 cm). **Grab** samplers are best used for preliminary mapping of the surficial sediments and for gross bulk chemical analysis.

**Parameters** common to most mine sites involve a visual description (physical logging) of the sediment, measurement of the wet to dry weight ratio, analysis of the particle size distribution, total organic carbon ( % loss on ignition), sulphides, ammonia, and metal concentrations. Consideration should be given to adding sediment pH and Eh (Redox State) measurements, as well as acid volatile sulphide concentrations, to the parameter list.

### *Decommissioning*

Monitoring should be completed to determine whether the sediments are continuing to act as a sink and burying the contaminants; or whether the pathway has reversed, with the sediments acting as a source of contaminants to the water column. Sediment sampling, in conjunction with a water-quality program, can be used to monitor contaminant loadings and identify any flow reversal of contaminants.

## Investigative Monitoring Techniques

A serious drawback to total bulk contaminant monitoring is the inability to correlate total contaminant concentrations to bioavailability and hence the potential for ecological effects. For this reason total bulk contaminant monitoring should only be used as a Tier I screening tool with Tier II or III sediment components initiated when predetermined "threshold concentrations" have been exceeded or when evidence of sediment related biological effects (e.g. in benthic macroinvertebrates) is detected. Additional Tier II or III sediment monitoring options that may be used to address questions of contaminant bioavailability are sediment **pore water chemistry** and **trace metal phase distribution** analyses either through **sequential extraction** or **acid volatile sulphide** relationships.

## 3.8 Section Bibliography

**AMD Studies Involving Sediment**

BC AMD 1991b
BC AMD 1990d
Brumbaugh et al. 1994
Herr and Gray 1996
Jones 1986
MEND. 2.11.1c 1992
MEND. 2.11.2a. 1993
Peine and Pieffer 1996
Smith and Macalady 1991

**Sediments as a Resource Exploration Tool**

Coker and Dunn 1983
Hornbrook and Garrett 1976
Sopuck et al. 1980

**Sediments and Contaminants: Reviews**

Fairchild et al. 1987
Forsten and Wittmann 1979
Forstner 1987
Hakanson 1983
Hart 1982
Reynoldson 1987

**Bioturbation**

Peterson et al. 1996
Reible et al. 1996

**Sediment Quality Criteria**

Ankley et al. 1996b
Chapman 1989
Chapman et al. 1987
CCME 1991, 1995
MOE 1993a

**Recommended Texts**

Horowitz 1991a
Mudroch and Azcue 1995

**Correcting Bulk Sediments For Particle Size**

Ackerman 1980
de Groot et al. 1982
Horowitz and Elrick 1988
Horowitz 1991b
Loring 1990

Loring and Rantala 1992

**Trace Element Chemistry - Physical Factors**

Ackermann et al. 1983
Filipek and Owen 1979
Forstner 1982b
Horowitz 1991a,b
Jenne et al. 1980
Mudroch 1984
Thorne and Nickless 1981
Sakai et al. 1986

**Trace Element Chemistry - Chemical Factors**

Jonasson 1977
Jones and Bowser 1978
Forstner 1982a,b
Horowitz and Elrick 1988
Hirner et al. 1990
Horowitz 1991a,b

**Study Design / Station Selection**

Baudo 1990
Env. Can. 1994
Håkanson 1984, 1992
MacKnight 1994
MOE 1993b
Mudroch and Azcue 1995
Mudroch and MacKnight 1991
Rowan et al. 1995

**Integrated Monitoring**
**(e.g. Sediment Quality Triad)**

**Sampling Methods and Equipment**

MEND. 4.1.1. 1989
MOE 1993b
Mudroch and Azcue 1995
Mudroch and MacKnight. 1991
Munch and Killey 1985
Norris 1988
NUTP 1985
Plumb 1981
Tetra Tech. 1986a
U.S. EPA. 1985b

**Sediment pH and Eh**

Bates 1981
Berner 1981
Mudroch and Azcue 1995
Thorstenson 1984

**Trace Metal Cycling and Bioavailabilty**

Forstner 1987
Hart 1982
Luoma 1983
Luoma 1989.
Salomons et al. 1987
Tessier and Campbell 1987
Thompson et al. 1984

**Sequential Extraction**

Chao 1984

Chapman 1986
Chapman et al. 1987, 1991, 1992
Green et al. 1993
Long and Chapman 1985
Reynoldson 1995

**Acid Volatile Sulphide (AVS)**

Allen et al. 1993
Ankley 1996
Ankley et al. 1996a,b
Besser et al. 1996
Carlson 1991
Chapman 1996
Di Toro et al. 1996a,b
Gonzalez, A.M. 1996
Hansen et al. 1996a,b

**Field AVS Investigations**

Ankley et al. 1996c
Hare et al. 1994
Liber et al. 1996
Mackey and Mackey 1996

**Laboratory AVS Investigations**

Di Toro et al. 1992
Petersen et al. 1996
Sibly et al. 1996

**AVS: Collection/Analytical Procedures**

Lasorsa and Casas 1996
Brumbaugh and Arms 1996
Leonard et al. 1996

Diks and Allen 1983
Greubel et al. 1988
Horowitz et al. 1989
Kheboian and Bauer 1987
Tessier and Campbell 1987
Tessier et al. 1979, 1982

**Quality Assurance / Quality Control**

BC AMD 1992
Brumbaugh and Arms 1996
Env. Can. 1994

**Within Station/Strata Replication**

Alldredge 1987
Brouwer and Murphy 1994
Cohen 1977
Green 1979, 1989
Hakanson 1985
Env. Can. 1991, 1995

**Sediment Pore Water**

Adams 1994
Adams et al. 1980
Azcue et al. 1996
Bufflap and Allen 1995a,b
Carignan et al. 1985, 1994
Davison et al. 1994
Hesslein 1976
Howes et al. 1985
Kromm et al. 1994
Rosa and Azcue 1993
Schults et al. 1992

## 4.0 COMMUNITY MONITORING

The impact of pollutants (e.g., metal contamination and/or acidification) on individual species may influence their population dynamics, and if severe or persistent can ultimately alter the community composition in the contaminated ecosystem. Obvious changes in community composition result when contaminants are acutely toxic to all or specific species in the community. Significant changes in community composition can also result from more subtle contaminant influences. For example, contaminant exposure on two competing species may alter the competitive interaction between the two species[18], resulting in a long-term shift in community composition and structure.

The objective of a community monitoring program is to detect and track these taxonomic changes through temporal and spatial (with respect to distance from point source) sampling of the community. The data are analyzed for discernible temporal and spatial patterns using a number of univariate biological indices and/or multivariate statistical procedures (ChIV:4.2.2.6) to examine the variation in the community composition and structure.

Often the terms "composition" and "structure" are used interchangeably in the literature; however, there is a distinct difference between the two terms. **Community Composition** can be defined as the presence/absence and abundance of taxa within a defined area (i.e., species list and densities). **Community Structure** is defined as the numerical description or pattern of the community composition data, for example the number of organisms, taxonomic richness, and species diversity.

**Indicator species** are commonly used to support investigations on community composition and structure. Indicator species are defined as:

- "a species or species assemblage (either taxonomically defined or fictionally defined such as a guild) that has particular requirements with regard to known physical and chemical variables such that changes in presence/absence, numbers, morphology, physiology, or behavior of that species indicate that physical or chemical variables are outside its preferred limits" (Rosenberg and Resh 1993).

---

[18] The competitive interaction may be altered in a number of ways. For example, species A may be negatively effected or the effect may be neutral on A but positive on B, etc.

The **presence** of an indicator organism/group is interpreted as evidence that a certain range of environmental conditions are being met. However, the **absence** of indicator organisms does not necessarily indicate that impacts on its physical and chemical environment have occurred. The absence of an indicator organism may be the result of geographic variables, competitive exclusion, or life-cycle events; this may indicate the need for further investigation. The following have been proposed (Rosenberg and Wiens 1976 and Hellawell 1986 in Johnson et al. 1993) as characteristics for the *ideal* indicator:

- taxonomic soundness and easy recognition by the nonspecialist. *Taxonomic uncertainties complicate long-term monitoring and between-site interpretation*;
- cosmopolitan distribution. *Allows for comparative studies on regional, national, and international scales;*
- numerical abundance. *Simplifies sampling and the identification of quantitative distribution patterns;*
- ecological characteristics well known;
- low genetic and ecological variability. *Often associated with narrow ecological demands, hence easier to define environmental requirements;*
- large body size (relatively speaking). *Facilitates collections and sample sorting;* and
- suitable for use in laboratory or micro/mesocosm studies. *Improves chances of determining cause and effect if impact detected.*

## 4.1 Selecting an AMD Community Monitoring Component

As briefly discussed in ChII:2.2.2.3 it is recommended that a community monitoring component be included in Tier I of the biological monitoring program. A number of aquatic communities may be used for community monitoring purposes including:

- fish communities;
- algal communities (planktonic algae and periphyton);
- zooplankton communities; and
- benthic macroinvertebrate communities.

The most significant drawback to top-down strategies such as community monitoring is that impacts are only detected after they have been expressed at the community level (ChII:2.2.2.3). This provides a mine operator with very little response time to investigate and mitigate contaminant impacts to that community. Hence, the community selected for monitoring purposes must be one which possesses a strong potential for recovery (e.g., recolonization, large reproductive potential, etc.) after mitigation and one not considered an economic resource. Of the four communities mentioned, fish communities are the least favourable in meeting these considerations. In the majority of situations, after the fact detection of significant impacts to fish populations and communities carries too high a cost in terms of negative regulatory and public reactions.

Mine tailings, AMD and other sources of metal contamination have been shown to alter algal community composition (Foster 1982; Crosseley and La Point 1988; Clements 1991, Rosassen 1997). However, there are a number of difficulties associated with using algal communities for Tier I monitoring of AMD, including:

- an imperfect knowledge of the life history and ecology of most species;
- poor taxonomic descriptions which make species identification difficult;
- wide fluctuations and very patchy distributions under natural conditions requiring intensive sampling and multiple reference stations;
- the ability of normally sensitive taxa to develop metal tolerance or form resistant spores;
- confounding factors arising from the close interaction between the zooplankton and phytoplankton communities; and
- the confounding interactions arising from the sensitivity of phytoplankton to nutrient additions which commonly accompany mining effluents.

For these reasons planktonic communities are considered to offer limited scope for the detection of toxic effects of aquatic contamination (Phillips and Rainbow 1993) and are not recommended as **generic** primary (Tier I) AMD monitoring components (BC AMD 1990, 1993). However, phytoplankton may prove to be an appropriate community-monitoring component on a site-specific basis.

Benthic macroinvertebrates are recommended as the Tier I AMD community monitoring component for most mining operations. In addition to their generic value for monitoring aquatic impacts (see ChIV:4.2), they are ideally suited for use in a tiered AMD monitoring program. Tier I monitoring of the benthic invertebrate community secures the benefits of community monitoring, while still providing enough response time to investigate any evidence of impacts to fishes. Monitoring of this community also monitors one of the indirect routes (food availability and contaminant transfer) for AMD impacts to fish populations, as they are significant prey items for many fish species. Tier I benthic macroinvertebrate community monitoring also serves to provide a biotic component to directly support the Tier I sediment contaminant program. The combined monitoring of sediment chemical composition, sediment benthic macroinvertebrate communities and the proposed Tier II sediment toxicity assays (ChIV:6.0) provides an integrated monitoring design incorporating concepts from the sediment triad (Chapman 1986; Chapman et al. 1991) and BEAST (BEnthic Assessment of SedimenT) (Reynoldson et al. 1995) approaches.

For additional literature on the effects of metal contamination on the other previously mentioned aquatic communities, see the section bibliography.

## 4.2 Benthic Macroinvertebrates

Invertebrates that inhabit the bottom substrates (sediments, debris, logs, macrophytes, filamentous algae, etc.) of aquatic habitats for at least part of their life cycle, and are retained by mesh sizes ≥ 200 to 500 • m are referred to as benthic macroinvertebrates (Rosenberg and Resh 1993). Benthic macroinvertebrates have and continue to be the most commonly used organisms for biomonitoring purposes. The advantages and disadvantages associated with using benthic macroinvertebrates and their communities as biomonitoring tools are summarized below, from the more detailed discussion provided by Rosenberg and Resh (1993).

***Advantages***

- Ubiquitous organisms that can be affected by many types of environmental perturbation in a number of different habitats.
- Large number of species allows for a broad range of responses to environmental stressors.
- Sedentary or limited individual ranges allow for spatial analysis of pollutant distribution.
- Long life cycles compared to other groups (e.g., phytoplankton) allows assessment of temporal changes caused by perturbation.
- Direct contact with water and sediment allows for assessment of biological responses to water and sediment quality.
- Important component in aquatic food chain.
- Well suited for experimental approaches to biomonitoring (especially for combining microcosm, mesocosm, and field community studies) .

***Disadvantages***

- Do not respond to all types of stressors and, therefore, must be balanced with other biological components.
- Contagious distribution of benthic macroinvertebrates requires a large number of samples for precise quantitative assessments.
- Sample processing and identification can be costly and time consuming[19].
- Distribution and abundance can be affected by factors other than water-quality (i.e., flow and substrate).
- Seasonal variation needs to be accommodated in the design of biomonitoring programs.
- Difficulty in identification of some groups.

Several studies have indicated the sensitivity of the benthic macroinvertebrate community to changes in metal concentrations (Winner et al. 1980; Clements et al. 1988; Clements et al. 1992, Nelson and Roline 1996) and pH (Dermott 1985, Fjellheim and Raddum 1990, Hall 1990), the two primary concerns arising from acid mine drainage. The response of the benthic community to AMD can be seen in the loss of sensitive groups of species, sometimes resulting in a reduction in the number of taxa present and/or a decrease in the diversity of organisms inhabiting the benthic regions.

Since benthic macroinvertebrate responses to AMD may vary, there are a number of different approaches which may be employed when using these organisms for monitoring purposes. These consist of investigations using:

1) community composition and structure;
2) indicator species; and
3) sentinel organisms.

The remainder of this section concentrates on developing monitoring programs based on interpreting the dynamics of the composition and structure of the benthic macroinvertebrate community, with additional support provided by the use of indicator species. The use of benthic macroinvertebrates as sentinel organisms for tissue contaminant loadings has primarily been developed employing marine organisms and/or for organic contaminants. The use of freshwater bivalves and large crustaceans

---

[19] Costs associated with benthic macroinvertebrate analyses are comparable to chemical analytical expenses for water, sediment and tissues incorporating metals and few additional parameters.

(e.g., crayfish)) for AMD biomonitoring is a field that merits further investigation (Tessier et al. 1984, Czarnezki 1987, Alikhan et al. 1990, Phillips and Rainbow 1993).

The use of benthic macroinvertebrate community composition and structure as indicators of change in environmental conditions is well documented in the literature ( Rosenberg and Resh 1993). Much of the work on developing indicator species has also employed benthic macroinvertebrates. The selection of benthic macroinvertebrate indicator species or assemblages may be accomplished by:

- studying the distribution of species among community types (pre-operational baseline data) followed by the classification of sites by macroinvertebrate assemblages with associated abiotic characteristics (Johnson and Wiederholm 1989); and
- further refined by selecting species of intermediate abundance, as rare species may be rare for several reasons other then pollution effects, and abundant organisms may have opportunistic characteristics promoting their abundance, rather than tolerance to pollution (Pearson et al. 1983)

Several studies related to elevated metal concentrations have shown trends in response patterns of specific groups of macroinvertebrates, primarily mayflies, caddisflies, and midges. Clements et al. (1988) compared field observational collections with controlled experimental studies and found that in both situations certain groups responded to elevated metal concentrations in predictable patterns.

- Ephemeropterans (mayflies) and chironomids (midges) of the sub-family Tanytarsini tended to be highly sensitive to increased metal exposure (Note Ch IV:4.2.2.5).
- Orthocladiini chironomids were more tolerant to heavy metal exposure.
- Similar patterns observed by other researchers (Winner et al. 1980; Wiederholm 1984) suggest that these groups contain species assemblages of value as indicator species for heavy metal pollution.

Designing the Tier I AMD benthic macroinvertebrate monitoring program follows the environmental plan discussed in Ch II:2.0. The three main components for development of the benthic macroinvertebrate monitoring program include:

1) environmental characterization;
2) **development** of the monitoring program and completion of the **intensive** environmental survey; and

3) commencement of **operational /decommissioning** monitoring. These are discussed in the following sections.

### 4.2.1 Environmental Characterization

Prior to the development of an operational benthic macroinvertebrate monitoring program, a pre-operational baseline characterization of the physical characteristics of potential benthic macroinvertebrate stations should be completed. This would be considered a part of the habitat classification as described in TSN 4.3 . Sediment particle size and hydrological characteristics are the most significant abiotic factors to be considered when developing benthic macroinvertebrate monitoring programs. Hence, data from the preliminary hydrological, sediment, and aquatic habitat surveys should be incorporated into the benthic macroinvertebrate preliminary survey.

***Minimum Requirements***

Habitat classification mapping should be completed ( TSN 4.3 ) confirming that the following information is available for all potential benthic macroinvertebrate stations.

- minimum physical characterization of the benthic macroinvertebrate habitat at potential stations.

| Substrate | Hydrology |
|---|---|
| - particle size | - water depth |
| - moisture content | - current velocity / wave exposure |
| - organic content | |

- classify sites as erosional or depositional

Areas where these characteristics are similar should be flagged as potential benthic macroinvertebrate sampling locations. This should result in the tentative identification of a number of potential reference station(s) and exposure stations. The exposure stations should be located along a spatial gradient from the predicted point of contaminant release.

A preliminary benthic macroinvertebrate sampling program should be completed at the potential operational sampling locations. This would involve the collection of replicated samples. This information would be used for:

- identification of rare, or potentially sensitive organisms and habitats;
- identifying any organisms (mayflies, chironomids, clams, crayfish) that would be useful as indicator species or sentinel organisms; and
- power analysis to determine the number of replicates required to meet the predetermined level of precision.

### 4.2.2 Monitoring Program

Benthic macroinvertebrate assessments are typically either part of the compliance or diagnostic monitoring objectives. The major decisions in the development of a benthic macroinvertebrate monitoring program are addressed in this section.

#### 4.2.2.1 Station Locations

Stations may be located in depositional or erosional zones of lakes and rivers. There are pros and cons to sampling either of these zones, though site-specific characteristics often dictate site selection (e.g. very few erosional zones in streams with low elevational relief).

***Lakes***

In most instances, benthic macroinvertebrate stations in lakes should be located in depositional zones for two reasons:

1) depositional sites are the areas of long-term contaminant accumulation; and
2) by locating the stations at the same sites as the sediment stations, correlations of sediment quality (i.e. contaminant load) may be made to the biological community.

***Rivers/Streams***

The selection of depositional (pools) versus erosional zones (riffles) in flowing waters is less obvious. The more transient nature of sediment depositional zones in flowing systems makes them less suitable for sediment and benthic macroinvertebrate stations, relative to lakes. The fauna in depositional zones of flowing water also tend to be more difficult to identify (e.g., chironomid populations) and less abundant relative to the fauna of erosional zones.

Benthic macroinvertebrates in erosional zones of rivers and streams are primarily exposed to water-borne contaminants rather than accumulated contaminants in the sediment. By locating the stations at the same sites as water-quality stations, correlations of water-quality may be made to the biological community. Conversely, benthic invertebrate depositional stations located in rivers and streams should be spatially correlated with sediment sampling stations.

### *Station Selection*

Once the general sampling habitat (erosional or depositional) has been determined, decisions regarding specific station locations are required. Station selection criteria are similar to those for selecting sediment stations; hence, less detail is provided here (see Ch IV:3.4.2.1). The results of any plume-delineation investigations or modelling (see Ch IV:1.1.2.3) should be consulted to assist in identifying station locations and mixing zones.

The following points should be considered when selecting impact or effluent-exposed stations.

1) Combine stations with sediment and/or water-quality stations when possible;
2) Stations should possess **similar physical and ecological characteristics** in order to minimize the potential for differences in community structure resulting from natural confounding factors (see following subsection). This should be confirmed using the data collected in the preliminary survey and habitat classification.
3) The first impact station should be located within the margin of the mixing zone nearest to the point of contaminant release (confirm using plume delineation). If the mixing zone does not commence until some distance downstream, then the nearest station should be located at the first suitable downstream habitat known to be within the contaminant plume.
4) Locate additional stations within the modelled mixing zone at increasing fixed intervals downstream. An effort should be made to locate these stations at geometrically increasing intervals (i.e., X, 2X, 4X, 8X) downstream, as detailed for sediment stations (Ch IV:3.4.2.1).

A minimum of one reference station is required, though, as previously discussed, multiple reference stations greatly enhance the power of a monitoring program by decreasing the chance of statistically detecting non-existent impacts (Ch II:2.2.2.4).

In addition to the first two points listed above for exposure stations, the following points should be considered when locating reference stations.

1) Upstream station(s) are preferred though these should be located neither too far upstream of, nor too close to, the contaminant point source. The former increases the likelihood of potential differences in biotic and abiotic variables (confounding factors). The latter raises concerns of indirectly influencing reference community composition through contaminant effects on mobile species such as fish, which may have been, or are exerting, a top-down influence on the benthic community (Env. Can. and DFO 1993).
2) If neighboring waterbodies (lakes or streams in adjacent drainages) are used as reference stations, they must be located at sites with similar biotic and abiotic features. When using different waterbodies as reference stations it is even more important to employ a multi-reference study design (Section 2.2.2.4).
3) For lake studies, reference stations located in the same bay as influenced stations are preferred, if it can be demonstrated that they are not exposed to the contaminant. Otherwise, adjacent bays with similar biotic and abiotic characteristics should be selected. Once again, a multi-reference study design is preferred.
4) The addition of a downstream station located at a site beyond the point of potential environmental effects[20] should be considered in order to assess community recovery.

### *Confounding Environmental Factors*

Confounding factors consist of a range of biotic and abiotic environmental variables that influence the composition and structure of benthic communities completely independent of mining activities. Stations should be selected such that these potential confounding factors are minimized between all of the stations (both reference and influenced). Note that some of the potential confounding factors listed below may be effluent related (e.g., pH), and hence would be considered a treatment effect and not a confounding factor.

The following is a brief discussion of the most common confounding factors for benthic macroinvertebrate community investigations. Many of these factors are inter-related (e.g., sediment particle size and current velocity); however, for the sake of simplicity they will be discussed as separate factors. These factors are all components which would have been parameterized during the

---

[20] Defined in the Environmental Effects Monitoring Guidelines for the pulp and paper industry as beyond the point where the effluent contribution has been diluted to < 1% (Env. Can.and DFO 1993).

Environmental Characterization Phase as part of the general habitat characterization as discussed in TSN 4.3.

### *Current Velocity and Discharge*

Current velocity affects the benthic community both directly and indirectly. Current velocity directly affects organisms through its physical influence on the organisms' mobility and capability to maintain position within the current[21]. Current velocity indirectly influences Species composition is indirectly effected by current velocity via the influence of current (turbulent or laminar flow) on oxygen concentrations and sediment deposition rates and particles size distribution.

Discharge is a confounding factor that should be standardized when a reference station is located in a different waterbody. It should also be determined during each sampling event as its effect on contaminant dilution rates may prove to be a confounding factor when comparing between sampling periods that experienced different discharges.

### *Depth*

Sampling depth indirectly influences the species composition of a benthic community in a number of ways. Water depth has a significant influence on light penetration as it pertains to periphyton and rooted macrophyte production, sediment deposition, and temperature and dissolved oxygen concentration resulting from stratification (ChIV:2.1.2). Standardization of monitoring stations by depth is of key importance in monitoring lake benthic macroinvertebrate communities.

### *Substrate*

The physical characteristics of the substrate is one of the most important factors influencing the characteristics of the benthic macroinvertebrate community (Klemm et al. 1990). All sampling stations (reference and exposed) should be standardized according to substrate composition.

### *Dissolved Oxygen*

---

[21] e.g., "clingers" such as some ephemeropterans (mayflies), and simulids (blackflies) can maintain position and forage in very high current velocities.

Benthic macroinvertebrates vary greatly in their dissolved oxygen requirements; hence, this is a significant factor in determining the species composition of the benthic community at a site. Visually similar habitats may possess different dissolved oxygen concentrations due to the influences of neighbouring habitats. For example, a pool at the base of a long deep run will have lower dissolved oxygen concentrations than one at the base of a highly oxygenated riffle zone. To minimize the influence of species specific oxygen requirements, stations should be selected which possess similar oxygen regimes.

*Tributaries*

Care should be taken when locating additional stations downstream of tributaries. The water-quality of the tributary may differ significantly from that of the waterbody under investigation. Differences may be due to anthropogenic influences, or natural influences resulting from the surface geology of the tributaries' drainage basin (e.g., natural acid drainage). The basic water chemistry of any tributaries should be determined during the Environmental Characterization Phase (Ch II:2.1.3.3), prior to finalizing the location of the operational monitoring stations.

***Accurate Relocation***

The ability to relocate a benthic macroinvertebrate station accurately is essential for repeated sampling through time. Information on the available options for siting in a station location for future sampling events is provided in TSN 9.0 .

#### 4.2.2.2 Sample Replication Requirements

To be of any value to a monitoring program benthic macroinvertebrate sampling **must** be replicated at each sampling station, during each sampling period. Most benthic marcoinvertebrate community monitoring programs are designed to test the null hypothesis ($H_o$) that there is no difference between community characteristics (e.g., abundance, diversity, etc.), either spatially (amongst the station(s)) or temporally (between sampling periods). Since benthic macroinvertebrates tend to exhibit highly contagious distributions (e.g., community parameters vary even within the immediate area designated as a sampling station), knowledge of **within**-station variability is required if differences **among** stations are to be confidently identified. Therefore, once the location of a sampling station has been determined, the number of sample replicates required to obtain a measure of within-station variability must be determined.

Statistical determination of within-station replication for monitoring programs is addressed by Green (1977) with discussions specific to benthic macroinvertebrates provided by Allan (1984), and Resh and McElravy (1993).

Considerations for sample replication:

- a **minimum** of three to five replicates should be collected for each station; and
- ideally, power analysis should be completed on the preliminary survey data to identify the number of replicates required to meet the predetermined level of precision (e.g., Appendix B). If the number of replicates is strictly determined by the budget, statistical analysis should be completed to identify the level of precision provided by the selected replication.

#### 4.2.2.3 Sample Timing

In most cases a fixed sampling frequency is appropriate for monitoring benthic macroinvertebrates. Sampling should be timed to approximate periods of maximum invertebrate diversity and should not be completed within a month of any significant freshet (Env. Can. and DFO 1993). In temperate climates, the period of maximum community maturity and richness occurs in early spring or late autumn; hence these are the most appropriate sampling periods (Cuffney et al. 1993). Winter sampling has the benefit of monitoring the population during potentially stressful conditions of low flows and food availability. However, winter sampling often presents difficult logistical problems. Sampling periodicity may range from annually to every 3 years depending on site-specific concerns.

#### 4.2.2.4 Selection of Sampling Equipment

A number of factors are involved in the selection of specific sampling devices and protocols. Standard benthic macroinvertebrate sampling equipment and methods should be used to collect **quantitative** samples. The two key equipment considerations involve determining whether depositional samplers, erosional samplers or artificial substrates are preferred and the mesh size to be used for sieving collections.

***Samplers***

For *in situ* monitoring purposes, samplers which collect benthic macroinvertebrates from the natural bottom sediments are preferred over samplers collecting organisms in the drift or colonizing artificial substrates (DFO and Env. Can. 1995b). Benthos collected from natural substrates represent the community formed through historical and current influences (DFO and Env. Can. 1995b). Organisms that have colonized artificial substrates, and especially those organisms in the drift, are more representative of short-term influences and poorly represent historical influences.

In depositional habitat, such as lake bottoms or slow moving areas of streams, grab or core samplers may be used. The selection of a depositional sampler for benthic macroinvertebrate monitoring is primarily based on the composition of the substrate being sampled; hence, the information provided on selecting sediment sampling equipment in the previous section (Ch IV:3.4.3.7) should be consulted for additional information.

**Grab samplers** are recommended over core samplers for monitoring the benthic macroinvertebrate community. Grab samplers have spring-loaded or gravity-operated jaws that close to entrap a certain volume of unconsolidated sediment (clay, silt, sand, etc.) and the organisms residing in the sediment. Figure IV:3.4-8 and Table IV:3.4-3 summarize the characteristics of the most common grab samplers along with recommendations for their use in specific substrates. Additional information related to benthic macroinvertebrate sampling is provided in TSN 4.2. Two of the most commonly used grabs for benthic macroinvertebrate studies are the Ponar and the Ekman grab samplers (Figure IV:3.4-5).

**Core samplers** (Table IV:3.4-4 and Figures IV:3.4-6 and 3.4-7) may also be used for collecting benthic macroinvertebrates from unconsolidated sediments. However, they are not used as much as grab samplers due to the small surface area they sample (Env. Can. and DFO 1993).

Benthic macroinvertebrate sampling in erosional zones, such as riffles in rivers, is best conducted with stream-net or **erosional samplers** (Figure IV:4.2-1). These samplers use mesh bags to sieve organisms from water flowing through the mesh. A summary of the advantages and disadvantages of the most common erosional (stream) samplers is shown in Table 9.2-1 with additional information provided in TSN 4.2.

Two commercially available and commonly used erosional samplers are the Surber sampler and various versions of cylinder samplers. While these samplers are relatively quantitative when used with care by an experienced biologist, they are sensitive to individual operator consistency. Variations of the Hess cylinder or invertebrate box sampler are recommended over Surber samplers as they are somewhat less sensitive to loss from beneath the sampler and from backwashing (Table 9.2-1). However, Surber samplers may now be equipped with base extensions to minimize loss beneath the sampler and to improve stability.

**Artificial substrates** are of use when sampling cannot be standardized for natural habitat features, or when it is not logistically possible to sample the natural communities (e.g., cobble substrate in deep water). Rivers and streams dominated by rubble and boulder substrates (e.g., common to Shield regions) are also best sampled using artificial substrates. Artificial substrate samplers are constructed of natural or artificial materials that are placed in water for a predetermined period for colonization by indigenous invertebrate communities (Env. Can. and DFO 1993). The Multi-plate (modified Hester-Dendy) and the basket sampler are the two most commonly used artificial substrate samplers. Examples of these samplers are shown in Figure IV:4.2-1, with additional information provided in TSN 4.2.

For more detailed discussions on sampling equipment, consult the review document by Klemm et al. (1990) and the references provided in the section bibliography.

### *Mesh Size*

Benthic macroinvertebrate samples require sieving to separate the invertebrates from fine sediment and debris. Once collected, samples should be field-washed using a maximum mesh size of 500 • m as proposed in the Canadian Environmental Effects Monitoring Guidelines for the pulp and paper industry (Env. Can. and DFO 1993). This is similar to the U.S. EPA recommendation (Klemm et al. 1990) of field washing through a U.S. Standard No. 30 sieve (approx. 600 • m). Compared to depositional samplers, erosional samplers tend to collect fewer fines and debris; hence, smaller mesh sizes may be used more efficiently. When comparing to historical databases the mesh sizes should be standardized or at a minimum accounted for during the interpretation of the databases. Mesh size should always be reported in the methods section of all reports.

### *Preservation, sorting and subsampling procedures.*

Samples should be fixed **after** field sieving using 10% formalin. Once fixed in formalin they may be stored as is, or washed and transferred to 70 - 80% ethanol (usually done in the laboratory). Samples to be used for biomass determination should not be preserved in alcohol due to the significantly greater weight loss associated with alcohol preservation compared to formalin (Klemm et. al. 1990).

Sorting and subsampling procedures as proposed in the Canadian Environmental Effects Monitoring Guidelines for the pulp and paper industry ( Env. Can. and DFO 1993) are recommended. Similar procedures are also detailed by Klemm et al. (1990) as recommended U.S. EPA protocols.

### *Field Notes*

Upon sample collection, both internal (pencil) and external (side of container, not lid) labeling should be completed. Each label should include the date, station code, and replicate number. Any large samples sub-divided into additional jars should be labeled as such (e.g., jar 1 of 2). The preservative type (formalin or ethanol) should also be detailed.

The following additional supportive data should be collected and recorded in the fieldbook.

- The above mentioned label information.
- The applicable habitat information such as substrate type, current velocity, water depth, and macrophyte composition.
- For streams, any changes in channel characteristics relative to the previously completed habitat characterization (see TSN 4.3).
- Field water-quality and limnology data such as temperature, pH, conductivity, dissolved oxygen, and Secchi disk transparency (Ch IV:2.1.3.1). Include type and size of field equipment operated.

### 4.2.2.5 Taxonomy

The taxonomic evaluation of benthic macroinvertebrate sample should be completed to the lowest taxonomic level confidently achievable (Env. Can. and DFO 1993). For most benthic groups this means identification to genus with some identified to species level. The desired identification levels for the major taxonomic groups are provided in Table IV:4.2-2. To meet these taxonomic requirements a qualified taxonomist is required and the creation of a reference collection for each site should be considered.

Generally, the present consensus supports this intensive level of taxonomic identification. The argument is that the use of higher taxa containing numerous diverse species can result in an environmentally heterogenous population being treated as a single non-representative variable. For example, ephemeropterans (mayflies) and trichopterans (caddisflies) are two groups often classified as sensitive indicators and are predicted to decline in abundance in response to decreasing water-quality. However, on a species specific level, tolerances can vary significantly. This is evidenced in work by Norris et al. (1982), who found that certain species of Ephemeroptera and Trichoptera were highly tolerant of trace metal pollution.

### 4.2.2.6 Data Analyses

Details on data analyses procedures for benthic macroinvertebrate investigations are beyond the scope of this document. However, there are a number of factors that are relatively unique to community databases that will be briefly addressed, with literature sources for more detailed discussions provided in the section bibliography.

#### *Unknown Immature or Damaged Organisms*

The standard-data handling and data-screening procedures outlined in Ch II:2.7 should be followed. Benthic macroinvertebrate databases often contain a number of organisms at different taxonomic levels classified as *Unknown* or *Unidentified*. These are usually organisms too immature or damaged to be identified to the next lower taxonomic level. These unidentified taxa and their abundances should be recorded in the raw databases; however, there are three options to be considered for handling these data during analysis (DFO and Env. Can. 1995b).

1) Apportion the number of *Unknown* individuals according to the ratio of identified individuals within that taxonomic level.
2) Delete these *Unknown* individuals completely from the analyzed database.
3) Complete the data analysis at the most detailed taxonomic level that has no organisms classified as *Unknown*.

Option 1 assumes that the ratio of *Unknown* individuals is similar to the ratio of identified individuals. This assumption is confounded by differing requirements for structural integrity for taxonomic identification (e.g., sensitivity of specific key morphometric characteristics to damage or developmental age), and differing life histories between species influencing the age distribution of the various species present.

With Option 2 there is the risk of severely biasing and misrepresenting the analysis if the *Unknown* component consists of a large proportion of taxa that account for a high proportion of the total benthic community. Option 3 has the same effect and drawbacks as restricting the taxonomic classification to lower hierarchical levels as previously discussed in Ch IV:4.2.2.5.

The most appropriate option is likely to be project-specific and determined by an experienced ecologist. The option selected should be clearly stated with a brief discussion of the basis for selecting that option.

### *Analysis Procedures or Techniques*

A wide range of analytical techniques or procedures is used in benthic macroinvertebrate studies, with a great deal of emphasis historically placed on univariate indices. While commonly used, the value of univariate indices of community structure (diversity, dominance, richness, etc.) has been in question for sometime (Hurlbert 1971; Washington 1984; Suter 1993). Diversity indices assess the numbers of taxa present (taxa richness) and the homogeneity of their numeric distributions in the community. Recently, the taxa richness component of diversity indices has been found to be more informative in demonstrating a metal gradient within receiving waters than the complete diversity index (Winner et al. 1980; Leland et al. 1989). However, changes in taxonomic richness may only represent short-term community responses, with species replacement masking any long-term changes in richness. Therefore, sole use of these measures/indices of community structure is not recommended.

Assessment of community structure through the combination of community composition data and multivariate analyses can provide more valuable information. The EEM guidelines for the pulp and paper industry (DFO and Env. Can. 1995b) recommend that the primary data investigative procedures for benthic macroinvertebrate monitoring programs include:

- an examination of variation in community structure based on multivariate taxa abundances and presence/absence, and
- an examination of variation in total benthic abundance and total richness (number of taxa).

The EEM guidelines outline the specific methods (e.g., canonical correspondence analysis, ANCOVA, and MANOVA) which may be used. Additional discussions on data analysis specifically for benthic macroinvertebrates are provided by Klemm et. al. (1990), and Norris and Georges (1993). Additional statistical references are listed in the section bibliography.

### 4.2.3 Pre-Operational Intensive Baseline

The purpose of the intensive baseline investigation is to collect detailed, replicated data on the benthic macroinvertebrate community at fixed points in time (pre-impact, specific sampling season) and space (the monitoring stations) against which future surveys (operational monitoring program) will be compared. This will result in a BACI study design, which, when combined with multiple reference stations, would provide the most rigorous study design (ChII:2.2.2.4). The intensive baseline provides a final opportunity for the *a priori* identification of potential indicator species (Pearson et al. 1983) and determination of the optimal number of stations and replicates required for the operational monitoring program.

**Minimum requirements**

- Match the protocols and stations of the proposed **operational monitoring program**.
  - i.e. sampling techniques, sampling season, etc.
- Consider additional replicates or sample stations (reference and impacted) to support any future decisions to intensify the monitoring program.
  - to minimize costs, these could be stored (appropriately preserved) and processed at a later date as required.
- Consider storing reference specimens in anticipation of the continued evolution of taxonomic classifications.
  - e.g. chironomid taxonomy has developed markedly over the last 10 to 15 years.

### 4.2.4 Operational Monitoring

The operational monitoring program should be executed as designed (see ChIV:4.2.2), incorporating any fine tuning developed during the completion of the intensive baseline. At a minimum, operational monitoring should include benthic macroinvertebrate collections for assessment of changes in community composition and structure. The data should be analyzed in order to determine whether there is evidence of marked changes in benthic macroinvertebrate population and/or community characteristics which can be related to mining activities. Sentinel organisms are best sued as investigative monitoring tools.

### 4.2.5 Decommissioning

The objective of benthic macroinvertebrate monitoring during the decommissioning phase is to determine whether recovery is occurring in any previously impacted areas. A key area will involve monitoring recovery of the benthic macroinvertebrate community on subaqueous tailings or waste-rock disposal sites. The section bibliography provides additional sources specifically concerned with the re-establishment of the benthic macroinvertebrate community on subaqueous disposal sites. An important component of the decommissioning baseline should involve sediment benthic macroinvertebrate toxicity assays for sediments known to be contaminated. This is discussed in greater detail in ChIV:6.1.

## 4.3 Quality Control

Along with the standard QA/QC procedures outlined in ChII:2.6.1, the following procedures should be incorporated into the benthic macroinvertebrate monitoring program:

- Ensure that the preservative is evenly distributed throughout the sample.
- Resort ten percent of all the samples to assess sorting consistency.
- Check a minimum of 10% of all sample splits during sorting or subsampling.
- Archive a reference collection.
- Recheck databases for accuracy, since benthic macroinvertebrate databases are highly susceptible to recording and transcription errors.

## 4.4 Investigative Monitoring Techniques

Previous sub-sections address the most common approaches and procedures used when incorporating benthic macroinvertebrate community analyses into a mine-monitoring program. In the following sub-sections some additional techniques are discussed which may be incorporated into the benthic macroinvertebrate program.

### 4.4.1 Functional Processes

Assessing the relationship between benthic macroinvertebrates and ecosystem processes is not a new technique; however, it is worth mentioning here as a potential monitoring technique. The benthic community functions as the major pathway for decomposition within aquatic ecosystems. Through the decomposition process, dead plant and animal material is broken down into compounds which can be absorbed by plants and used in primary production. This provides a key loop for the recycling of nutrients within the aquatic system. Decomposition involves bacteria, fungi, and benthic invertebrates.

Environmental perturbations, such as elevated metal concentrations and/or acidity, can disrupt the processing of detrital material in the ecosystem through impacts on the benthic invertebrate community. These changes in the benthic community can reduce the efficiency of the decomposition process and thus cause increased accumulation of organics and inorganics within the receiving waters.

There are two relatively simple methods available for assessing changes in ecosystem processes.

- Leaf pack litter bags (Petersen and Cummins 1974; Reice 1980).
  - place a known biomass of leaf litter (or macrophyte) in a porous bag
  - position bags within the reference and exposure study areas
  - remove bags at specified intervals and determine remaining biomass
  - rate of change in biomass is used as a quantitative value of decomposition rates.

- Functional classifications of benthic macroinvertebrates (Merritt and Cummins 1984)
  - incorporate functional classifications into the analysis of the community composition.
  - functional classifications are based on the morphology of mouth parts.

Shredders - ingest Coarse Particulate Organic Matter (CPOM)
Collector/Filterers - ingest suspended Fine POM
Collector/Gatherers - ingest FPOM from surface films
Scrapers - ingest attached algae and associated material
Predators - ingest living animal tissue
Parasites - feed on living animal tissue without directly killing the host

From an ecological perspective, the assessment of both benthic community structure and benthic processes, such as decomposition, can provide valuable insight into the health of the entire ecosystem. Assessing both structure and process provides the opportunity to understand the significance of species changes within the system, and monitor how these changes disrupt the natural processes of the ecosystem.

## 4.5 Section Summary

The objective of a community monitoring program is to detect and track taxonomic changes through temporal and spatial (with respect to distance from point source) sampling of the community. A community monitoring component should be incorporated into Tier I of the biological monitoring program.

For most mine sites **benthic macroinvertebrates** are likely to be the best Tier I AMD community monitoring component. However, this should be determined on a site-specific basis as other communities may be more suitable in certain circumstances. The combined monitoring of sediment chemical composition, sediment benthic macroinvertebrate communities, and the proposed Tier I/II sediment toxicity assays (ChIV6.2.2) will provide an integrated monitoring design.

### *Environmental Characterization*

Areas with similar physical characteristics should be identified during habitat classification mapping as potential benthic macroinvertebrate sampling locations. A preliminary benthic macroinvertebrate sampling program should be completed at the potential operational sampling locations for:

- identification of rare, or potentially sensitive organisms and habitats;
- identifying organisms useful as indicator species or sentinel organisms; and
- power analysis to determine the number of replicates required to meet the predetermined level of precision for the operational monitoring program.

### *Developing the Operational Monitoring Program*

In most instances, benthic invertebrate stations in **lakes** should be located in depositional zones. The availability of suitable erosional (i.e., cobble-gravel substrate) or depositional (i.e., organic material) sites determines the appropriate sampling substrate for **streams** and **rivers**. Erosional sites are generally preferred as they tend to have greater invertebrate abundances and/or community diversity. Macroinvertebrate stations located in erosional zones should be paired with water-quality monitoring stations, while those located in depositional zones should be paired with sediment and water-quality stations.

Stations should possess **similar physical and ecological characteristics** to minimize confounding factors. The first impact station should be located within the margin of the mixing zone nearest to the point of contaminant release. Additional stations should be established at increasing intervals downstream, with the last downstream station located in a depositional site considered to be beyond the point of potential environmental effects.

Reference/Control station(s) should be located at sites identified in the extensive survey as being similar to the impact stations. Multiple control or reference sites/waterbodies are preferred. If suitable reference sites are not available, then the program is restricted to a gradient study design.

It is common for three to five **replicate samples** to be collected at each station; however, the most appropriate procedure is to complete statistical power analyses on preliminary sampling data. Sampling should be timed to approximate periods of maximum invertebrate diversity and should not be completed within a month of any significant freshet. In temperate climates, sampling in **early spring** or **late autumn** usually meets these criteria. Sampling periodicity may range from annually to every **3 to 5 years** depending on site-specific concerns.

For *in situ* monitoring purposes, samplers which collect benthic macroinvertebrates from the natural bottom sediments are preferred over samplers collecting organisms in the drift or colonizing artificial substrates. **Grab samplers** are recommended over core samplers for community monitoring in depositional zones, with **erosional net samplers** used in erosional habitats. **Artificial substrates** are of use when sampling cannot be standardized for natural habitat features, or when it is not logistically possible to sample the natural communities.

Sorting and subsampling procedures as proposed in the Canadian Environmental Effects Monitoring Guidelines for the pulp and paper industry (Env. Can. and DFO 1993, DFO and Env. Can. 1995b) are recommended. **Taxonomic identification** should be completed by a qualified taxonomist to the lowest taxonomic level confidently achievable with a reference collection archived.

## Pre-Operational Intensive Baseline

The purpose of the intensive baseline investigation is to collect detailed, replicated pre-impact data on the benthic macroinvertebrate community against which future surveys (operational monitoring program) can be compared. The protocols and stations should match those of the proposed operational monitoring program, though reference specimens and additional samples (if desired) may be stored pending future needs.

### Operational Monitoring

The operational monitoring program should be executed as designed, incorporating any fine tuning developed during the completion of the intensive baseline. At a minimum, operational monitoring should include benthic macroinvertebrate collections for assessment of changes in community composition and structure. The data should be analysed to determine whether there is evidence of marked changes in benthic macroinvertebrate population and/or community characteristics which can be related to mining activities.

### Decommissioning

Benthic macroinvertebrate monitoring during the decommissioning serves to measure recovery rates in previously impacted areas. Sediments identified as contaminated by the decommissioning baseline should be periodically reassessed using sediment-benthic macroinvertebrate toxicity tests.

### Investigative Monitoring Techniques

#### *Functional Processes*

From an ecological perspective, the assessment of both benthic community structure and benthic processes, such as decomposition, can provide valuable insight into the health of the entire ecosystem. Two relatively simple methods may be used for assessing changes in ecosystem processes. These include **leaf pack litter bags**, and the incorporation of **functional classifications** into the analysis of the benthic macroinvertebrate community composition.

## 4.6 Section Bibliography

### Indicator Organisms: Theory/ Reviews

Hellawell 1986
Johnson et al. 1993
Pearson et al. 1983
Rosenberg and Wiens 1976
Schubert 1984

### Benthic Invertebrates and Metal Contamination

Alikhan et al. 1990
BC AMD 1990b
Chadwick et al. 1986
Clements et al. 1988, 1992, 1994
Czarnezki 1987

### Benthic Invertebrates and Biomonitoring: Reviews

Rosenberg and Resh  1993

### Phytoplankon Monitoring/Metal Effects

Clements  1991
Crosseley  and La Pointe  1988
Foster  1982
Parsons et al  1986
Price and Morel  1984
Rai et al.  1981
Rosassen  1997
Stokes 1974, 1983
Stokes  et al. 1973
Yan  1979
Whitton  1984

### Fish Community Monitoring

Adams et al.  1992
Angermeier  1995
Faush et al.  1990
Lester et al.  1996

### Zooplankton Community Monitoring

Yan and Strus  1980

### Sample Replication

Allen  1984
Ferraro et al.  1989
Morin 1985
Schwenneker and Hellenthal  1984
Resh and McElravy  1993

Leland et al.  1989
Nelson and Roline 1996
Norris  1986
Norris et al.  1982
Tessier et al.  1984
Winner et al.  1980

### Benthic Invertebrates and pH

Dermott  1985
Fjellheim and Raddum  1990
Hall  1990

### Artificial Substrates

Casey  1994
Flannagan and Rosenberg  1982
Rosenberg and Resh  1982

### Study Design

Cuffney et al.  1993
DFO and Env.  Can.  1995b
Env.  Can.  1985
Env.  Can and DFO  1993
Klemm et al.  1990

### Interpretation and Analyses of Invertebrate Data

Chadwick and Canton  1984
Clarke  1993
Faith et al.  1991
Norris and Georges 1993
Resh and Jackson 1993

**Sampling Equipment**
Downing 1984
Klemm et al. 1990
Peckarsky 1984
Storey and Pinder 1985

**Taxonomic Considerations**
Cranston 1990
Env. Can and DFO 1995b
Furse et al. 1984
Herricks and Cairns 1982
Osborne et al. 1980
Resh and McElravy 1993

**QA/QC and Sample Processing**
Cuffney et al. 1993
DFO and Env. Can 1995b
Klemm et al. 1990
Suter 1993
ter Braak and Verdonschot 1995
Washington 1984

**Ecosystem Functioning Approach**
Allard and Moreau 1986
Fairchild et al. 1987
Hildrew et al. 1984
MacKey and Kersey 1985

## 5.0 SENTINEL COMPONENT - ADULT FISH INVESTIGATIONS

Fishes may exhibit an acute or chronic response to AMD contaminants. The source compliance and diagnostic water-quality monitoring programs (ChIII:2.0 and ChIV:2.0) and effluent toxicity assays (ChIV:6.0) should serve to prevent acute AMD effects on the fish population[22]. Chronic AMD effects are more difficult to monitor and may influence fish either directly or indirectly as shown in Figure IV:5.2-1.

Indirect effects arise when AMD impacts food sources or habitat which in turn influences the health of the fish, or alters the total carrying capacity of the waterbody (Munkittrick and Dixon 1991). Direct effects occur when exposure to AMD contaminants has toxicological implications to the fish. Through time, one or both of these factors will influence population size and structure.

---

[22] With the exception of catastrophic spill or accidental release events.

The most important AMD constituents that may influence fish are pH and mobilized trace elements. Reductions in the pH of aquatic ecosystems are known to affect behavioural (e.g., avoidance responses), respiratory and other physiological fish functions. Reduced pH also has a serious effect on recruitment in a number of different fish species (Harvey and Jackson 1995, Jackson and Harvey 1995). Generally, early life stages are the most vulnerable to reductions in pH (Harvey 1982). This can lead to the loss of year classes and, hence, alteration in the age structure of the population. Eventually the most sensitive species may disappear from the fish community. Mobilized trace elements (e.g., aluminum, copper, nickel) can similarly affect behaviour, recruitment and/or lead to the bioaccumulation of these contaminants in juveniles and adults.

Due to the importance of fishes as an economic and recreational resource, and the sensitivity of regulatory and public opinion to impacts to the fishery, it is recommended that some form of fish monitoring be incorporated into the Tier I AMD monitoring program.

### 5.1 Environmental Characterization

A fisheries assessment should be completed in both exposed and reference waterbodies as part of the Environmental Characterization Phase. A comprehensive species list should be developed from historical information and a preliminary sampling program[23]. The relevant biological and habitat requirements should be identified from the literature for each of the species present on the species list.

Complete habitat classification and mapping (TSN 4.3) with an emphasis on habitat required for dominant, economically important and rare fish species identified in the species list. The following critical fish habitat should be identified and rated:

- fish spawning habitat;
- rearing habitat (species and possibly life-stage specific);
- foraging habitat;
- overwintering habitat;
- fish migration routes; and
- natural (e.g., beaver dams) and anthropogenic barriers to fish movement.

---

[23] Non-fatal sampling methods should be used whenever possible (See TSN 4.4).

Information on species-specific habitat requirements for the more common/economically important Canadian fishes may be found in the references listed in the section bibliography. Special attention should be paid to those portions of the waterbodies likely to receive point source (e.g., proposed effluent release points), non-point source (e.g., stack releases, site disturbance effects such as sediment inputs) pollutants, stream road crossings, and culvert installations. For information on road crossings and culvert requirements, see the references in the chapter bibliography.

## 5.2 Fish Monitoring Options

Fish are used as a monitoring component for most mining operations in Canada. They have been used to varying degrees for monitoring purposes at all organizational levels shown in Figure IV:0-1. Fish monitoring requirements will differ depending on the sensitivity of the population to sampling stress, and the risk of AMD impacts. Several fish monitoring options are discussed in the following paragraphs.

### 5.2.1 Population Monitoring

The combined direct and indirect effects of a contaminant(s) may be monitored at the organism, population or community levels[24] of organization (Figure 9.0-1). As previously discussed (ChIV: Intro), community level monitoring of fish populations provides inadequate response time to protect fishes. Monitoring fish at the population level increases the chances of detecting an effect while it is still reversible and before it is reflected at the community level in the form of local species extinction. A population's size and age structure are reflections of the sensitivity of population processes such as recruitment, and age-specific fecundity[25] mortality rates, to the surrounding environment, and hence can prove to be sensitive to direct or indirect contaminant exposure (McFarlane and Franzin 1978, Munkittrick and Dixon 1989a,b).

A full-scale monitoring program designed to accurately determine the population dynamics of even a single fish species requires comprehensive information on the population age/stage structure, and age/stage-specific survivorship, and reproduction rates for both exposed and reference sites. At a minimum, this would require a sampling program incorporating seasonal sampling in a number of

---

[24] A community consists of a number of interacting populations. A population consists of a number of interacting individuals of the same species.

[25] An individual's potential reproductive capacity based on mature germ cell (e.g. eggs) production.

habitats, as habitat use can vary both seasonally and with developmental stage. The value of such an intensive population investigation as an operational monitoring component should be carefully considered due to concerns of cost efficiency, difficulties in separating natural fluctuations from AMD effects (BC AMD 1992), and the potential of sampling mortality causing more serious impacts than chronic contaminant exposure. This is especially of concern in northern oligotrophic waters where fish growth-rates are slow.

#### 5.2.1.1 Simplified Sentinel Population Monitoring

A simplified monitoring program for determining individual level characteristics which strongly influence or correlate with population processes may be used as a proxy for the more intensive population studies. It has been proposed (Munkittrick 1989a,b, BC AMD 1990, 1992, Env. Can. and DFO 1993, DFO and Env. Can. 1995a), that the monitoring of individual organism characteristics (traditional condition indices) of a **sentinel** fish species may serve as a fish surveillance program. This would require data on individual indices which reflect reproductive potential (e.g., No. eggs per female, gonad weight, age at maturity), growth (length and weight as a function of age), and energy stores (liver weight and condition factor). Additional data on sample population parameters such as mean age, and time to maturity, would also be required. Such an approach has several advantages, including:

- the use of traditional fisheries techniques (individual fish measurements);
- requires less intensive sampling of fish communities and populations; hence is cost effective and less destructive; and
- may be focussed on a limited age/size class of the population, which also minimizes sampling costs and mortality.

The objective of such a monitoring program is the early detection of pseudo-population level effects in the sentinel species and the preliminary identification of the individual level parameters being influenced (i.e., growth, reproduction, or survival). This allows Tier II and III investigative procedures to be focussed towards the specific component being influenced. For example, should the Tier I program provide evidence of recruitment failure, then Tier II investigations may be focussed on reproductive success and larval mortality investigations.

### 5.2.2 Visual Fish Health Assessments

The addition of an in-field fish health assessment to the sentinel fish monitoring program would enhance information return without significantly increasing costs or sampling effort. Chronic stress related to pollution, including pH and trace metal exposure, can lead to a myriad of conditions that generally serve as indicators of decreased health. As fish are often the most visible part of aquatic ecosystems, the dramatic visual effect of fish diseases and developmental abnormalities can raise public concerns in excess of the direct implications to fish population dynamics. The benefits of monitoring the occurrence of diseases and abnormalities in exposed and reference waters are threefold.

1) Disease outbreaks and some abnormalities are natural events; hence, failure to monitor occurrences in reference waters can lead to the mining operation being incorrectly identified as the cause[26].
2) The incidence rates may be used as an indicator of potential chronic effects, prior to significant population or community impacts.
3) Early detection prior to significant outbreaks provides response time for operators to launch investigative programs on their own initiative (proactive approach) rather than in response to regulator or public pressure.

Commonly used health indicators are briefly discussed in the following subsections.

### 5.2.2.1 Disease Due to Pathogens, Neoplasms and Parasitism

One of the more visible results of chronic contaminant exposure is an increase in the incidences of fish disease outbreaks, possibly due to chronic stress impairment of the immune system. Increased incidences of neoplasms[27] has also been evidenced with metal contaminant exposure (Hinton 1993, Baumann et al. 1996). Trace metal exposure can enhance (McFarlane et al. 1986, Robohm 1986) or impair the immune systems ability to cope with infectious agents and uncontrolled cell proliferation (Wolf 1988, Leighton 1996). Increased and/or decreased parasitism burdens have also been linked to pollution effects (Wedemeyer and McLeay 1981, Poulin 1992). Increased parasite burdens may result from chronic stress impairment of the fishes' immune system (Poulin 1992). Alternatively, contaminants may decrease fish parasite burdens either directly through effects on the parasite itself

---

[26] This is further exacerbated by the lack of information on natural disease outbreaks. Most fish disease investigations have focussed on hatchery and contaminant-exposed populations.

[27] Abnormal proliferation of cells which distort the shape of normal tissue.

(especially the free-living stages) or indirectly by reducing the population of the parasites' intermediate hosts.

### 5.2.2.2 Skeletal and Developmental Abnormalities

Skeletal anomalies, such as fin and skull deformities, the absence of one or more fins and sometimes the complete pelvic girdle, shortened operculae, fused and deformed vertebrate and spinal curvature, have all been associated with pollution exposure (Sloof 1982), along with congenital defects, parasitic infections, and electrical shocks (e.g., lightning and electroshocking) (BC AMD 1990). Contaminant induced spinal curvature tends to increase in frequency with long-term exposure (BC AMD 1990). Metal contaminants (AMD and non-AMD related) have also been shown to act as teratogens[28], causing deformations involving the eyes, heart, skeleton, and pigmentation (Munkittrick and Dixon 1989b, Weis and Weis 1991).

### 5.2.2.3 Tissue Related Indices

Contaminant exposure can also result in the enlargement or atrophy of specific fish tissues, depending on the metabolic response to the contaminant (BC AMD 1990). Liver enlargement occurs with increased enzyme activity associated with the detoxification processes. Atrophy of gonads can arise from food limitation or reproductive dysfunction (BC AMD 1990). Organosomatic indices measuring the ratio of organ weight to body weight (or length-specific body weight) have been used as an indicator for these scenarios (Goede and Barton 1990). The most commonly used organosomatic indices are the hepatosomatic (liver) index (HSI) and the gonadal somatic index (GSI) (Anderson and Gutreuter 1983).

A healthy fish tends to exhibit a specific relationship between total body weight and length, which is often used to assess the overall condition of the fish. The most common condition index is the condition factor, expressed as weight/length$^3$ multiplied by a scaling constant. A similar index is the relative weight index (Wr), where $Wr=(W/Ws)\bullet 100$. Ws is an empirically based length-specific standard weight, which has only been determined for some species (Wege and Anderson 1978).

A decline in the condition factor is often interpreted as evidence of the depletion of energy reserves (e.g., liver glycogen or body fat). This depletion of energy reserves may be due to a decrease in

---

[28] Contaminants which produce or induce deformities during development.

energy input (lower food availability and/or foraging efficiency) or increased metabolic requirements, both of which have been associated with contaminant related stress.

These indices are sensitive to a number of factors including sex, stage of sexual development, seasonal fluctuations, and species. These factors should be standardized if comparisons are to be made. Condition indices may be used as a screening tool in combination with other assessment measures.

#### 5.2.2.4 Health Assessment Index

An autopsy-based assessment process was proposed by Goede and Barton (1990) as a rapid, cost-efficient means of detecting changes in the health of fish populations. This was refined and simplified into a **H**ealth **A**ssessment **I**ndex (HAI) (Adams et. al. 1993) which quantifies the general health of the sample populations and allows statistical comparisons to be made between exposed and reference populations. The parameters and classification procedures are shown in Table IV:5.2-1, with case studies of the use of the HAI provided by Adams et. al. (1993).

The authors present the HAI as a reliable method for assessing the general health of a fish population, which has the advantages of being simple, rapid, and cost-effective. As such it would be classified as a Tier I monitoring component for screening purposes due to its limited diagnostic value. Experienced fishery personnel could incorporate the HAI into any fish monitoring programs with little additional cost or time expenditure.

### 5.2.3 Monitoring Identified Spawning Sites

The early developmental stages (egg and larvae) of fishes have been shown to be sensitive to AMD related contaminants (Munkittrick and Dixon 1988b, Wien and Wien 1991, Lovich and Lovich 1996). Monitoring of the early life-stages and of spawning adults, should be considered at mines where locally important spawning sites have been identified as at risk from AMD. This biological monitoring component should be integrated with a water-quality program design as detailed in ChIV:2.1.4.3. Such a scenario is found in British Columbia where AMD from the Mt. Washington mine drains into salmon and trout spawning habitat in the Tsolum River. At this site *in situ* egg bioassays are incorporated into the monitoring program every three years (BC Environment 1995).

The tendency for fish to exhibit avoidance responses to contaminants has implications for fish-spawning migrations. For example, spawning habitat located upstream of a point source release, while not directly affected by the contaminant, may be indirectly affected if reproductive adults are deterred from migrating upstream through the contaminant concentration gradient to spawning habitat. This may only be of concern if spawning habitat is a limiting factor for the local fish population. This underlies the need to complete a spawning habitat assessment for the greater study region to determine the **relative** importance of spawning habitat to the local fish population.

### 5.2.4 Monitoring Bioaccumulation in Fish

Freshwater fish species have been used as biomonitors of trace contaminants more frequently than any other taxon (Rainbow and Phillips 1993). The selection of fish as biomonitors for contaminant accumulation has primarily been in relation to health concerns over human consumption and has logically concentrated on the chemical analysis of fish muscle tissue. **However, such monitoring programs are inappropriate for AMD monitoring purposes where trace metals are the contaminants of concern**. In addition to the failure of most fish species to meet the standard recommendations for sentinel-bioaccumulation monitors, fish muscle tissue is an extremely poor indicator of trace metal exposure (BC AMD 1990). There is abundant evidence that levels of regulated metals (especially the common AMD components of Cu and Zn) do not concentrate in fish muscle tissue (Miller et. al. 1989, Rainbow and Phillips 1993).

The failure of fish muscle tissue to reflect AMD exposure was documented by Miller et al. (1989) in white sucker (*Catostomus commersoni*). These researchers found that white sucker exposed to AMD (Cu and Zn) showed no significant increase in muscle tissue, though liver, kidney and gill tissue did exhibit significantly increased metal burdens (also see Miller et al. 1992). Bone metal concentrations have also been found to conservatively reflect environmental concentrations (Bendell-Young et. al. 1986, Miller et al. 1989).

## 5.3 Generic Tier I Fish Monitoring Program

The following is a recommended Tier I fish monitoring program designed to meet the broadest range of monitoring requirements, in the most cost-efficient and least destructive (i.e., to the fish population) manner. The following protocols are primarily based on the papers by Munkittrick and Dixon (1989a, 1989b), and Munkittrick (1992), with additional information provided in BC AMD (1990), Env. Can. and DFO (1993) and DFO and Env. Can. (1995a). The reviews of the first round of Environmental Effects Monitoring (EEM) for the pulp and paper industry, currently in progress, should also be consulted when they are complete. **Protocols should always be refined to reflect site-specific circumstances**.

It is recommended that a minimum of one fish species be selected as a sentinel species and monitored every three to five years for individual parameters considered to have implications at the population level. To maximize the information return from this sample population, the opportunity should also be taken to complete a rapid fish health assessment (ChIV:5.2.1.2).

### 5.3.1 Selecting the Sentinel Species

The appropriate sentinel species should be identified on a site-specific basis employing the following characteristics proposed by Munkittrick and Dixon and (1989a) and Env. Can. and DFO (1993).

- A fish species with maximum exposure to the contaminant plume should be selected. The ideal fish species would be one with limited movement patterns that remained resident in the exposure zone through all of its developmental stages.
- A fish species from the lower trophic levels, such as a benthic forager, would reflect contaminant stresses with less of a time lag than upper trophic level piscivorous[29] fishes. A benthic forager would also integrate well with the sediment and benthic macroinvertebrate community monitoring programs.
- The selected species should be an integral component of the food web. This increases the probability that effects detected in the sentinel species will be transmitted to other ecosystem components.
- A species with an intermediate life-span (long enough to respond to chronic exposure, but short enough to reflect responses within a few sampling periods).
- A species with high energetic requirements is preferred as evidenced by a rapid growth rate, high fecundity and early age at maturity.
- Abundant species are preferred as they are often easier to sample and the population will be less sensitive to sampling pressure.

Smaller fish species (e.g., forage species) best conform to these characteristics. Note that neither the economic importance nor suitability for human consumption are determining factors. The species should be selected solely on its merit as a sentinel for monitoring ecosystem health.

---

[29] Fishes whose dominant prey items are other fish.

Site-specific AMD flushing patterns should also be considered when selecting species. Fish species exposed to peak AMD flushing events (ChII:1.4) during early developmental (egg and larvae) stages should receive special consideration, especially if exposure coincides with low periods of dilution.

Table IV:5.3-1 provides an example of a ranking scheme for selecting an appropriate sentinel species. Such a comparison process should be completed for each of the fish species identified during the Environmental Characterization Phase (ChIV:5.1).

### 5.3.2 Reference Population

The success of the fish monitoring program depends on the selection of a suitable reference population(s) for comparison purposes. Investigations are often restricted to a single reference population due to budgetary and time constraints, though this increases the chance of falsely interpreting natural differences between the populations as mine impacts (false positive). The habitat and the population characteristics of the reference population should be as similar to the exposure population as is feasible. To achieve this, intensive baseline information (ChIV:5.2.3) should be collected from a minimum of two additional populations, with the most similar selected as the reference population.

Reference populations may be located upstream from the point source release, in neighboring or tributary watersheds, in upstream or neighboring lakes, or in adjacent bays of large lakes (Env. Can. and DFO 1993). When using upstream reference populations it must be shown that movement between the reference and the exposed population is restricted (e.g., fish barrier such as a beaver dam). Several potential confounding factors should be considered when selecting a reference study area. These have already been outlined in ChIV: 4.2.2.1.

### 5.3.3 Sample Size

Munkittrick (1992) found that 15 to 25 white suckers from an appropriately selected reference site provided adequate information for a preliminary characterization of population averages. Increasing the sample size was found to decrease the standard error but did not significantly alter the mean. A minimum of 20 males and 20 females is recommended for EEM monitoring in the pulp and paper industry (Env. Can. and DFO 1993). Ideally, the data from the intensive baseline should be used to determine the appropriate sample size for the species selected. Examples of the appropriate analytical procedures for determining the required sample size can be found in DFO and Env. Can. (1995a).

### 5.3.4 Parameters of Interest

The number of sentinel species as well as incidental species captured per collection location should be recorded. It should be possible to identify the specific net and mesh size that each fish is captured in.

The following information should be recorded from each of the sentinel fish (20 of each sex).

- Sex
- Length and weight (record for all fish captured)
- Liver and Gonad weight
- Gonad Maturation Phase (Table IV: 5.3-2)
    1) immature
    2) developing (green), ripening, or maturing
    3) mature, running ripe, or spawning
    4) spent or recovery phases

The following **structures/organs** should be collected from each fish.

- Gut contents
    - Large piscivorous gut contents may be recorded in the field.
    - Otherwise collect and preserve (10% buffered formalin) the foregut.
    - Determine relative abundance of prey items.
- Gonads
    - Collected from gravid or pre-running-ripe fish for egg count and size determination.
    - Preserve ovaries in 10% formalin or a modified Gilson's fluid (Snyder 1983)
- Ageing structures
    - Species specific age structures are shown Table IV: 5.3-3.

In many instances otoliths have replaced scales as the preferred ageing structures, hence, recent literature should be consulted for the selected sentinel species. If ageing techniques are not well documented for the selected sentinel species, have not been recently updated (approx. last 5 yrs), or if an old, slow-growing population is suspected, then otoliths should be collected (Env. Can. and DFO 1993).

A cursory **visual fish health assessment** should be completed for each fish.

- At a minimum, document fish condition using the data sheet shown in Table IV: 5.3-4.
- Photograph evidence of tumors, neoplasms, and/or lesions of major organs.

Alternatively, the slightly more involved semi-quantitative autopsy approach previously discussed in ChIV: 5.2.1.2 and summarized in Table IV: 5.2-1 may be used.

Liver and kidney should be collected for **metal analyses** (TSN 4.5) for comparison between the reference and exposed population(s). As discussed in ChIV:5.2.1.4, liver and kidney total metal concentrations are currently recognised as more reliable indicators of metal exposure than metallothionein. However, in anticipation of improved techniques and a greater understanding of metallothionein induction (ChIV:5.2.7.5), it is recommended that liver and kidney metallothionein be collected as part of the intensive survey in conjunction with total metal concentrations in these organs.

### 5.3.5 Collection Periods and Methods

The appropriate sampling time must be determined on a site-specific basis. Fish that are resident in the contaminant plume year round may be sampled at any time of the year, though sampling[30] as fish begin to congregate for spawning may prove the most efficient approach. Site- and species-specific factors will need to be considered for sentinel species not present year round (e.g., spawn in a tributary) in the plume. Sampling is not recommended immediately after spawning due to the additional confounding factors of spawning recovery, resorbing of eggs, etc.

Fish collection permits should be obtained from the relevant fisheries regulatory body (usually provincial) prior to executing any sampling program. The appropriate sampling equipment will depend on the fish species, the target size range, stage and seasonal behavioural characteristics of the target species, and the type of aquatic habitat to be sampled. The most common fish-collection equipment is identified in Table IV: 5.3-5 with additional information provided in TSN 4.4 and references in the section bibliography. It should be noted that any investigations of relative abundance require standard sampling equipment, net set periods and protocols (see Nielson and Johnson 1983).

### 5.3.6 Data Analyses

The basic principles of this approach involve detecting patterns of changes in the measured parameters that identify with the range of population response patterns provided in Table IV: 5.3-6. These response patterns, as they apply to fish populations, are discussed in detail by Munkittrick and Dixon (1989a). The ability to detect a specific response pattern provides the investigator with a tool to determine appropriate Tier II and III follow-up investigations in a more time- and cost-efficient manner. For example, evidence of a “Type II-Recruitment Failure” directs the investigators to concentrate Tier II and III investigations on concerns related to deterioration of spawning or rearing habitat, or factors related to stress-induced spawning failures.

Details on specific data analyses procedures for a fish monitoring program are beyond the scope of this document. The papers by Munkittrick and Dixon (1989a,b) and the guidelines outlined by DFO and Env. Can. (1995a) should be consulted. The required analysis is relatively simple and does not

---

[30] If egg counts are to be determined, fish must be collected before they become too ripe, as eggs can be released and lost during handling. .

require the somewhat more advanced techniques used for the benthic macroinvertebrate community component.

### 5.4 Pre-Operational Intensive Baseline

A pre-operational intensive baseline investigation should be completed specifically focussing on the reference and exposure sentinel populations. The investigations should focus on determining the following information for both the exposure and reference population(s).

1) Identify and complete a detailed classification of spawning, rearing, foraging, and habitat for the sentinel species. Information collected as part of the baseline investigations for the sediment and benthic macroinvertebrate components may be used to gain a greater understanding of the fisheries habitat.
2) Confirm specific spawning sites, either through the actual observation of spawning events, the location of concentrations of fish in ripe reproductive condition in a suitable spawning habitat, or by egg collections from the habitat.
3) Complete a comprehensive fisheries investigation for the sentinel species. This requires sampling all of the habitats identified in point No. 1 using the appropriate sampling equipment. Length and weight (and sex if possible) should be determined for all of the sentinel species captured along with the completion of a brief external[31] health assessment. A few fish from a wide range of size classes (if possible)[32] should be sacrificed and processed as discussed in ChIV:5.2.2.4. This should also include determination of total metal concentrations for liver and kidney. The determination of metallothionein concentration in these tissues should also be considered in anticipation of future requirements.

The data collected for both the exposure and reference population(s) should be compared to confirm the suitability of the reference population. The causal factors for differences detected between the populations should be investigated and identified (e.g., differences in carrying capacity of the habitats) for consideration during future assessments. Of special interest is the accurate determination of the

---

[31] An external assessment for hemorrhaging, lesions, and external parasites can be completed without killing the fish.

[32] Sampling during spawning migrations restricts size classes to mature fish. The suitability of this should be determined on a site-specific basis.

average time to maturity for the proposed sentinel population. Sampling during the operational phase should be designed to concentrate on this portion of the population.

### 5.5 Operational Monitoring

The operational monitoring program should be executed as designed (ChIV:5.2.2), incorporating any improvements developed during the completion of the intensive baseline. To minimize impacts from the sampling program, the fish monitoring cycle should be no less than 3 years. Preliminary evidence of differences in response patterns between the reference and the exposure populations should be assessed at the next monitoring cycle by completing the Tier I investigations and possibly incorporating selected Tier II components. Evidence of significant mortality, severe changes in fish health, or disease outbreaks restricted to the exposure population should result in the immediate initiation of Tier II and III investigations. No specific recommendation may be made on the appropriate Tier II and III procedures as they will be site- and effect-specific.

### 5.6 Decommissioning

A Tier I fisheries monitoring component is not required during the decommissioning phase if there were no fisheries impacts during the operational phase. In this case fisheries investigations become Tier II/III monitoring components. Fisheries investigations would only be initiated if one of the other Tier I decommissioning monitoring components (e.g., such as water and/or sediment) exhibited a significant decline in quality.

Evidence of fisheries impacts in the operational phase would require that decommissioning incorporate a fish-monitoring program completed in conjunction with any mitigative or specific recovery programs. It is not possible to provide specifics as the recovery programs and monitoring requirements will vary depending on the specific impact.

### 5.7 Quality Control

Along with the standard QA/QC procedures outlined in ChII:2.6.1 and those provided in BC AMD (1992), the following standard field sampling information should be provided during each sampling event.

- Location of sampling areas and fish collection sites (mapped).
- Habitat description including potential confounding factors:

  - water depth, temperature, current velocity, dissolved oxygen, substrate classification, specific conductance, pH, visual evidence of pollution.
- Date and time of collection
- Names of field crew.
- Fish collection methods and specifics on equipment:
  - information required to determine catch per unit effort such as Net set times, mesh sizes, voltage for electroshocker, shocking time, etc.
- Photograph near shore sampling locations.

Upon completion of the field program, quality control procedures are required for fish ageing and taxonomic verification.

- Confirm a minimum of 10% of all fish ageing should be confirmed through blind submissions to a second ageing technician.
- Verify the taxonomic identification of any species-or age-classes difficult to identify (e.g., separating young-of-the-year (YOY) white sucker from YOY longnose sucker).
- Archive a reference collection of any taxonomic verifications.

### 5.8 Investigative and Developing Monitoring Techniques

The Tier II and III follow-up investigations often employ techniques considered too specific or experimental for use as general monitoring tools. The decision process for determining Tier II and III components is outlined in the following section. This is followed by a brief discussion of new and developing techniques which may be of use as Tier II or III components. These techniques are addressed in this section as the majority of the work has focussed on fishes. For a more detailed review of these developing techniques in relation to AMD, see BC AMD (1990, 1993).

### 5.8.1 Identifying Tier II and III Requirements

The objective of Tier II and III investigations is to confirm the impact detected by the Tier I program and to determine whether any cause-and-effect relationship between the mining activity and the effect exist. Selection of the appropriate Tier II/III investigative tools requires an integrated assessment of all the Tier I monitoring components (water and sediment quality, hydrology, and benthic macroinvertebrates) for evidence of links to the effect suspected in the fisheries component.

For example, the Tier I components should be examined to determine whether the observed fisheries effect is indirectly driven by alteration of habitat or food availability, or acting directly by toxicologically disrupting functioning of individual organisms. Habitat-alteration investigations require assessment of physical changes in fish habitat or investigations of fish avoidance responses. Gross alteration of this habitat can be quickly identified or ruled out. Continued use of previously identified (Environmental Characterization and Intensive Baseline) spawning and nursery sites and migration routes should be confirmed.

Should disruption in the functioning of individual organisms be suspected, then a wide range of investigative approaches and tools must be assessed. Due to the complexity and often experimental nature of these approaches they have only been briefly addressed here. For additional information, see the references in the section bibliography.

Many of these approaches have also been proposed as early warning indicators for Tier I or screening monitoring purposes. However, the use of these bottom-up monitoring tools has a number of drawbacks, as discussed in the introduction to Chapter IV. Currently, they are of more value as Tier II/III investigative tools. With further development these techniques will begin to play a greater role in the Tier I level of many monitoring programs.

### 5.8.2 Histopathology

Histological examinations should be considered a Tier II component for evaluating tissues shown to be abnormal in a large number of fish in exposure populations. Histopathology involves the microscopic study of tissues for evidence of histological changes. Toxic effects on biochemical and physiological systems are ultimately expressed as changes in cellular and subcellular morphology. The major organs of fishes in which histopathological responses to stress are seen include the skin, liver, gills, and kidney (Hinton and Laurén 1990).

One of the primary advantages of histological approaches is the ability to rapidly fix a number of tissues or whole fish (for small species) in the field for detailed examination in the laboratory at a later date. Histological examinations can identify specific organs, cells and organelles that have been affected and serve to focus the study towards additional physiological or biochemical investigations. They also indicate chronic effects in exposure populations.

The most serious disadvantage to histology as a monitoring tool is the difficulty in distinguishing anthropogenic toxicant effects from the effects of natural stressors, infectious diseases, or the normal range of physiological changes. A relatively large number of "normal" tissues are required to confidently assess an abnormal characteristic. There is also the difficulty of correlating histological alterations or clear abnormalities to notable population or community level changes. For a review of the role of histology as an indicator of stress in fishes, see Hinton and Laurén (1990).

### 5.8.3 Immunological Indicators of Stress

The immune response in fishes can be compromised by stress, whether natural or anthropogenic in origin. Many fish pathogens are ubiquitous in the aquatic environment or persist in fish which serve as asymptomatic carriers. Contaminant exposure can lead to immuno-suppression resulting in outbreaks by previously benign or managed pathogens. Hence, evidence of increased disease incidence or outbreaks in the Tier I monitoring program may require Tier II or III investigations incorporating tests for immuno-suppression.

There have been a great number of advancements in our understanding of fish immuno-functions and in the development of immuno-assays. This work has primarily been driven to meet the needs of fish farming and hatchery operations and is thus based on stressors unique to these operations. Our knowledge of anthropogenic-related immuno-suppression of wild populations is more limited. However, the commercial requirements for rapid immuno-assay techniques have resulted in improved techniques for investigating fish diseases and immuno-suppression. A review of assessment options and fish disease outbreaks is provided by Anderson (1990).

### 5.8.4 Molecular and Biochemical Indicators

Chronic exposure to low or moderate levels of stressors can lead to a wide range of alterations in an organisms' growth, reproduction, and disease resistance. All of the responses to stress are preceded by chemical changes in nuclei, cells, and extracellular fluids (Thomas 1990). It is argued that, because molecular and biochemical responses are the earliest indicators of contaminant exposure, they are especially useful as early warning indicators. However, the very sensitivity of these proposed indicators has made it difficult to identify the biological and/or ecological relevance of detected responses.

Investigations into molecular indicators (e.g., DNA adducts, or strand breakage, and molecular actions of hormones) are a recent development and few have been adequately characterized to permit their use as biological monitoring tools (Thomas 1990). Investigations involving biochemical indicators is somewhat more advanced. Metallothionein is one that has received a significant amount of attention and, hence, is discussed in greater detail in the following sub-section. A number of additional potential biochemical parameters have been investigated and are reviewed in NRCC (1985), Jimenez and Stegeman (1990), and Thomas (1990).

#### 5.8.4.1 Metallothionein

The protein metallothionein has been proposed as a potential indicator of heavy metal stress (Klaverkamp et al. 1984, 1991, Hamilton and Mehrle 1986, Klaverkamp and Duncan 1987; Malley et al. 1993, St. Louis et al. 1993). While metallothionein is not currently widely employed as a routine monitoring tool, it has recently been receiving a great deal of attention from regulatory bodies.

Metallothionein, or MT, binds to all of the group Ib and IIb metals (Cu, Ag, Au, Zn, Cd, Hg) with special affinity for cadmium, copper, mercury, and zinc. The primary function of MT is thought to be the homeostatic metabolism of essential metals such as copper and zinc (Hamer 1986), though there appears to be a secondary role involving heavy metal detoxification (Klaverkamp et al. 1984; Hamer 1986, Hamilton and Mehrle 1986). In fish, these metal binding proteins are primarily distributed within the liver, kidneys, gills, and intestines (Roesijadi 1992).

The potential of metallothionein and other metal binding proteins as biomonitoring tools is based on their ability to bind toxic metals, and the ability of certain metals to induce synthesis of these proteins. While this has been documented in some field applications (Farag et al. 1995, Hamza-Chaffai et al. 1995), others have found no significant increases in metallothionein concentrations at field exposure concentrations sufficiently high to show increased metal accumulation in the liver (Palace and Klaverkamp 1993). Unfortunately, many non-metallic factors can also induce metallothionein synthesis (Kaji and Kojima 1987). Metallothionein concentrations have been found to vary between fish species, as a function of reproductive status (increase during spawning), feeding status, and with size and geographic location within a species (Hamza-Chaffai et al. 1995).

At present, due to the lack of knowledge of the dose-response relationship for metallothionein synthesis and metal exposure, and the deficiency of information on metallothionein concentrations considered to be evidence of health impairment, it is not recommended that metallothionein be relied on as a sole bioindicator of fish health (Cope et al. 1994, Hamza-Chaffai 1995). While metallothionein may be used as a screening tool for evidence of metal exposure, measurements of total metal concentration in the liver and kidney are likely to be more reliable (e.g., see Palace and Klavercamp 1993) for screening and are easier and less time-consuming to collect.

## 5.9 Section Summary

Due to the importance of fishes as an economic and recreational resource, and the sensitivity of regulatory and public opinion to fisheries effects, it is recommended that the Tier I monitoring program incorporate a fish sentinel species, using monitoring tools operating at the whole organism level or lower.

### Environmental Characterization

A fisheries assessment should be completed in both exposed and reference waterbodies as part of the Environmental Characterization Phase. This assessment should include a comprehensive species list, estimates of relative abundance, and a detailed fish habitat assessment.

### Operational Monitoring Program

Fish monitoring requirements will differ on a site-specific basis depending on the sensitivity of the population to sampling stress, and the risk of AMD impacts. The basic recommended program involves surveillance monitoring of individual organism characteristics of a **sentinel** fish species. **Individual indices** which reflect reproductive potential, growth and energy stores are measured along with data on **sample population parameters** such as mean age, and determination of the time to maturity. This sentinel fish-monitoring program is further enhanced by incorporating an in-field autopsy-based **fish health assessment** without significantly increasing costs or sampling effort.

The objective is the early detection of pseudo-population level effects in the sentinel species and the preliminary identification of the individual level parameters being influenced (i.e., growth, reproduction, or survival). This Tier I program provides the information required to focus Tier II and III investigative procedures towards the specific component being influenced.

The appropriate sentinel species should be identified on a site-specific basis, though smaller fish species are recommended. Fish species exposed to peak AMD flushing events during early developmental (egg and larvae) stages should receive special consideration, especially if exposure coincides with low periods of dilution. A suitable reference population(s) of the same species occupying similar habitat and exhibiting similar population characteristics is required.

Ideally, the data from the Intensive Baseline should be used to determine the appropriate **sample size** for the species selected. The present recommendation for EEM fish sentinel monitoring for the pulp and paper industry is a minimum of 20 males and 20 females of the same species.

### *Parameters of Interest*

The sex, length, weight (total, liver, and gonad), and gonad maturation phase should be recorded for each fish, with gut contents, ovaries, and ageing structures collected for processing in the laboratory. The autopsy-based fish-health assessment is completed while the above measurements and structures are collected.

Tissues should be collected and analysed for evidence of **metal bioaccumulation**. Bone metal levels may be used to indicate chronic AMD exposure, with liver, kidney, and/or gill arches analysed to indicate recent metabolic activity related to metal exposure. In anticipation of future scientific and regulatory developments, the collection of liver and kidney metallothionein is also recommended as part of the intensive survey.

### Pre-Operational Intensive Baseline

A pre-operational intensive baseline investigation should be completed specifically focussing on the reference and exposure sentinel populations. The investigations should include the following information.

- Identify and complete a detailed classification of spawning, rearing, foraging, over-wintering, and habitat for the sentinel species. Suspected spawning sites should be confirmed by a spawning investigation.
- Complete a comprehensive fisheries investigation for the sentinel species. This requires sampling all of the habitats identified above using the appropriate sampling equipment.

### Execution of the Operational Monitoring

The time interval between fish-monitoring cycles should be no less than three years to minimize impacts from the sampling program. Preliminary evidence of differences in response patterns between the reference and the exposure populations should be assessed at the next monitoring cycle by completing the Tier I investigations and incorporating selected Tier II components. Evidence of significant mortality, severe changes in fish health, or disease outbreaks restricted to the exposure population, should result in the immediate initiation of Tier II and III investigations.

### Decommissioning

A fisheries decommissioning component is only required if fisheries impacts were detected during the operational phase, or if effects are detected in other components (e.g., water, sediments, etc.) of the decommissioning monitoring program.

### Identifying Tier II and III Requirements

The selection of Tier II and III fish monitoring components will be on a site- and effect-specific basis. An assessment of the other Tier I monitoring components (water, sediment, invertebrates, and fish) should be made to determine whether the observed effect on fish is operating **indirectly**, through alteration of fish habitat or food availability, or **directly** disrupting physiological functions. The results of this assessment should be used to identify the appropriate investigative approach, a few of which are briefly addressed below.

- **Histological** examinations should be considered as a Tier II component for evaluating tissues shown to be abnormal in a large number of fish in exposure populations, or when the suspected contaminant is known to result in histological alterations. The major organs of fishes in which histopathological responses to stress are seen include the skin, liver, gills, and kidney.
- Contaminant exposure can lead to immuno-suppression resulting in outbreaks by previously benign or managed pathogens. Hence, evidence of increased disease incidence or outbreaks may be investigated by incorporating **immunological indicators of stress** and tests for **immuno-suppression**.
- A range of **molecular and biochemical indicators** has been proposed as early warning monitoring parameters. However, the very sensitivity of these proposed indicators makes it difficult to identify the biological and/or ecological relevance of detected responses.

## 5.10 Section Bibliography

**Benthic Invertebrate (BI) Trace Element** **pH / Metal Effects on Fishes**

**Bioaccumulation**

Hare et al. 1991
Hare and Campbell 1992
Miller et al. 1992, 1993

**BI Gut Corrections**

Brooke et al. 1996
Hare et al. 1989
Phillips and Rainbow 1988

**Fish Trace Element Bioaccumulation**

Bendell-Young et al. 1986
Besser et al. 1995
Miller et al. 1989, 1992, 1993

**Handling Tissues for Chemical Analysis**

Crawford and Luoma 1993
Tetra Tech 1986c

**Monitoring Fish Immuno-Competence**

Anderson 1990
Blazer et al. 1987
Haaparanta et al. 1996
Petermann 1990
Rice et al. 1996

**Approaches to Fish Sentinel Monitoring**

BC AMD 1990, 1992
DFO and Env. Can. 1995a
Env. Can. 1995
Env. Can. and DFO 1993
Munkittrick and Dixon 1989b,c
Munkittrick 1992
Farag et al. 1995
Harvey 1982
Harvey and Jackson 1995
Hodson and Blunt 1986
Jackson and Harvey 1995
Lovich and Lovich 1996
McFarlane and Franzin 1978
Munkittrick and Dixon 1988a,b,c, 1989a, 1991
Weis and Weis 1991

**Monitoring Potential of Histology**

Haaparanta et al. 1996
Hinton 1993
Hinton and Laurén 1990

**Fish Tissue Indices**

Anderson 1980
Anderson and Gutreuter 1983
Wege and Anderson 1978

**Reproductive Indices and Fecundity Measurements**

Bagenal 1978
Chigbu and Sibly 1994
Donaldson 1990
Snydecor 1983

**Metallothionein**

Cope et al. 1994
Dutton et al. 1993
Farag et al. 1995

### Neoplasms and Contaminants

Baumann et al. 1996
Black 1984
Harshbarger et al. 1993
Hinton 1993
Hinton and Laurén 1990
Maccubbin and Ersing 1991
Premdas and Metcalfe 1995
Premdas et al. 1995
Smith et al. 1989

### Individual Level Fish Health Assessments

Adams and Ryon 1994
Adams et al. 1993, 1996
Goede and Barton 1990
St. Louis et al. 1993

### Fish Ageing

Brothers 1987
Casselman 1987
Jearld 1989
Thompson and Beckman 1995
Trippel and Harvey 1991

### Fish Habitat

***Walleye (Stizostedion vitreum)***

Auer and Auer. 1990
Busch et al. 1975
Corbett and Powles 1986
Geiling et al. 1996
Johnson et al. 1977
Kitchell et al. 1977

Hamer 1986
Hamilton and Mehrle 1986
Hamza-Chaffai et al. 1995
Kaji and Kojima 1987
Klaverkamp and Duncan 1987
Klaverkamp et al. 1984, 1991
Malley et al. 1993
Palace and Klaverkamp 1993
Prud'homme 1993
Roesijadi 1992

***Lake Trout (Salvelinus namaycush)***

Gunn 1995
Gunn et al. 1996
Johnson et al. 1977.
Marcus et al. 1984
Marshall 1996
Martin,, and Olver 1980

***Brook Trout (Salvelinus fontinalis)***

Raleigh. 1982

***Bull Trout (Salvelinus confluentus)***

Fraley and Shephard. 1989
Rieman and McIntyre 1993

***Pacific Salmons (Oncorhyncus sp.)***

Liaw 1991
McMahon et al. 1984
Rawson 1957

***Yellow Perch (Perca flavescens)***
Krieger et al. 1983.

***Northern Pike (Esox lucius)***
Casselman and Lewis 1996
Cook and Solomon 1987
Cook and Bergersen. 1988
Diana et al. 1977
Inskip 1982
Johnson et al. 1977
McCarrraher and Thomas 1972
Minns et al. 1990, 1996
Voellestad et al. 1986

***Suckers (Catostomus sp.)***
Edwards 1983a,c
Twomey et al. 1984

Minnows (Cyprinidae)
Edwards 1983b
McMahon 1982
Trial et al. 1983a,b

***Atlantic Salmon (Salmo salar)***
Beland et al. 1982

***Brown Trout (Salmo trutta)***
Raleigh et al. 1984b

*Bass (Micropterus sp.)*
Edwards et al. 1983
Johnson et al. 1977

Bjornn and Reiser. 1991
Collings 1974
Hale et al. 1985
Hickman and Raleigh 1982
McMahon 1983
Raleigh and Nelson. 1985
Raleigh et al. 1984a, 1986

***Whitefish***
**(*Coregonus sp. & Prosopium sp.*)**
Northcote and Ennis 1994
Quadri 1968

***Arctic Grayling (Thymallus arcticus)***
Hubert et al. 1985
Merkowsky 1989

**Sampling With Nets**
Hubert 1983
Craig et al. 1986
Pierce et al. 1990

**Piscicides**
Davies 1983

**Electrofishing**
Reynolds 1983
Hill and Willis 1994
Hayes 1983
Lyon 1992

Stuber et al. 1982

**Fish Culvert Requirements**
Katapodis 1992.

## 6.0 TOXICITY TESTING

The objective of this section is to provide operators and monitoring personnel with a basic understanding of toxicity tests and their role in the environmental management program of a mine. For additional information, the literature in the section bibliography should be consulted.

The standard use of toxicity testing has been to determine in the laboratory contaminant specific acute toxicity, to establish national and regional standards or surface water-quality guidelines. These laboratory assays also serve as a source monitoring components to assess the potential biological toxicity of effluent released to the receiving environment. A third role for toxicity testing involves its use as a Tier II or III investigative tool for determining cause- and-effect relationships for biological effects detected in the receiving environment.

There are two general approaches to toxicity testing:

1) controlled laboratory experiments; and
2) *in situ* natural ecosystem studies.

Current test methods for both approaches include single-species tests, multi-species tests and ecosystem tests:

- **Single-Species Tests**

  Single-species toxicity tests are the standard protocol for toxicity assessments and are typically performed in controlled laboratory settings. Due to the control over extraneous variables in a laboratory, cause-and-effect relationships can be established from single-species toxicity testing.

- **Multi-Species Tests**

These studies involve the use of laboratory microcosms or small-scale enclosures containing samples from the natural ecosystem. They incorporate several different species, typically fish, invertebrates and plants. The advantage of the multi-species tests over the single-species tests is the ability to detect changes beyond the level of one species, thus providing information more directly related to an ecosystem. Multi- species toxicity tests are not standard tests and are more typically used in research programs addressing specific questions.

- **Ecosystem Tests**

  Laboratory microcosms can be used to simulate ecosystems; however, ecosystem tests are more typically conducted in the natural environment through field experiments. Ecosystem tests can be performed on whole ponds/lakes or in a single lake subdivided into several test areas through the use of enclosures. Field ecosystem toxicity tests have the added advantage of providing data obtained from natural conditions. However, the variability of environmental parameters may cause some interpretation problems.

- **Mesocosm Studies**

  These studies bridge the gap between laboratory multi-species studies and field ecosystem studies. Mesocosm studies usually involve the use of multiple species established in artificial ponds and/or streams. Artificial streams have been used in assessments of AMD on stream biota (BC AMD 1990a, 1992). Recent research in the pulp and paper industry employed artificial stream mesocosms set up in the field to assess ecosystem/community responses to effluent inputs (Culp and Podemski 1995; Culp et al. 1996a,b). These designs provide replicated treatments and standardization of environmental variables and hence have excellent potential as Tier III investigative tools for determining cause and effect relationships. For a review of the use of artificial streams see Lamberti and Steinman (1993). For a recent discussion of the use of artificial streams for assessing effluents see Culp et al. (1996).

There are limitations to both laboratory and field toxicity testing designs as well as various advantages and disadvantages to using single-species, multi-species, and ecosystem approaches. With the exception of single species compliance effluent bioassays (see ChIV:6.1.1), the role of toxicity tests in monitoring is generally restricted to Tier II and/or III investigative monitoring.

## 6.1 Types of Toxicity Tests

Aquatic toxicity test methods can be categorized according to the length of exposure, test situation, criteria of effect, and organism type. Typically, effect criterion (endpoint), and exposure duration are used to describe the type of toxicity test.

### 6.1.1 Screening Toxicity and Toxicity Identification Evaluation (TIE)

Standard requirements for most mining operations are to conduct single-species screening toxicity tests of effluent to determine the toxicity prior to release into the environment. These tests are not used to determine the concentration of a causative agent; rather they serve as an indicator test (typically 50% mortality is the effect criterion). Screening toxicity tests should be conducted on the AMD at regular intervals. If the AMD is found to cause significant mortality to test organisms, then screening through Toxicity Identification Evaluation (TIE) should be performed in order to determine the causative agent(s). Once the causative agent has been determined, measures should be taken to reduce its concentration in the AMD.

TIE uses a chemical methodology in which chemicals are extracted from the test solution in a sequential order based on physical and chemical properties with each fraction screened for toxicity. A TIE should be conducted on effluents that have proven to be acutely toxic in a screening toxicity assessment. The results from these assessments will provide information required for proper treatment of the contaminated water or process alterations. Once the causative agent has been determined, information regarding its level of toxicity should be obtained either from the literature or through cause-effect toxicity tests such as an acute toxicity test.

### 6.1.2 Acute Toxicity

These tests evaluate the relative toxicity of chemicals to organisms over a short-term exposure period under controlled environmental conditions. The endpoint/effect criterion is either mortality, immobility, loss of equilibrium or some measure of desired effect (physiological or biochemical). Most acute toxicity tests are time-dependent, running either 24, 48, or 96 hours. Some tests may be time-independent, in which case the exposure period is not predetermined and may run as long as 21 days in duration. The results of an acute toxicity test are typically reported as the concentration of the chemical required to cause mortality (LC50) or another effect (EC50) in 50% of the test organisms.

Acute toxicity tests are not typically part of a standard monitoring program, but could be required by regulatory agencies if environmental impacts are evident in the receiving environment, or if information pertaining to the causative agent is not available in the literature.

### 6.1.3 Chronic Toxicity

A chronic toxicity test is designed to test several life stages of a test organism. The test organisms are exposed continuously to the test agent for a sufficient period of time to allow for growth, development, maturity, and offspring production. Test populations are exposed to several concentrations of the test agent and are observed at regular intervals for an assessment on development. Control tests are conducted simultaneously and all changes are compared with "normal" development of the control test organisms.

Chronic toxicity tests are not part of a standard monitoring program but could be used in Tier III special investigations. Assessment of long-term impacts of an acidic drainage event may require the use of chronic toxicity tests if field observations indicate long-term changes in species populations and/or community composition.

## 6.2 General Toxicity Testing Guidelines

A general summary of toxicity testing guidelines (primarily single-species tests) is provided in the following sections.

### 6.2.1 Water-Toxicity Testing

The use of standardized methods for conducting water-toxicity tests is essential to ensure uniformity and reproducibility. It is imperative that standard protocols be adhered to as effluent toxicity testing is generally a compliance requirement. Several agencies have produced standard methods for the toxicity testing of water pollutants (ASTM 1988a, 1988b; APHA 1992) and most laboratories follow these guidelines as closely as possible to ensure quality data. The basic requirements for a water toxicity tests are listed below.

- Maintain an adequate supply of dilution water that is acceptable to the test organisms and the purpose of the test. Natural uncontaminated water from the receiving environment should be used if possible, however, dechlorinated dilution water can be used if only source.
- Characterize each batch of dilution water prior to use. Analysis should include hardness, alkalinity, conductivity, pH, particulate matter, and total organic carbon.
- Sample test waters or effluents at the pipe prior to release to the receiving environment. The sample collected at this point is considered a 100% test solution.
- use test waters within 36 h after collection, unless it has been shown that toxicity does not change with time. If samples are not used within 2 h of collection time, they should be preserved by storing in the dark at 4°C.
- Create test concentrations by mixing test waters with dilution waters. Prior to mixing water, they should be agitated to ensure even distribution of contaminants. Test concentrations are dependent on the objectives of the study. For the purposes of the AMD monitoring program, it is suggested that 100% test solutions should be used in the screening toxicity test.
- Use test organisms that are a uniform age and size. Indigenous species from the receiving environment can be used, however, it is more conventional to purchase test organisms from a supplier. Typically water toxicity tests are conducted on the water flea (*Daphnia* sp.) or fathead minnow and rainbow trout larvae/fry. Proper care and handling of test organisms is essential.
- Ensure that proper experimental design is followed (dependent on study objectives).
  - *A priori* decisions include the dilution factor, number of treatments, and number of test chambers and organisms per treatment that are required to reach the objectives of the study.
  - For the purposes of the monitoring program, screening toxicity tests should be conducted using controls and a 100% test solution.
  - For acute toxicity tests, one or more control treatments and a geometric series of five concentrations of test material should be assessed
  - The test chamber is the experimental unit. As the number of test chambers per treatment increases, the degrees of freedom increases and the power of the hypothesis test increases. Typically a minimum of 6 test chambers per treatment are used (10 is preferred).
  - A randomized block design is preferable with each treatment being present in each block.
- Ensure that the test duration (10 day screening/acute tests, several week chronic tests) and selected endpoints (growth and survival for acute tests, others for chronic tests) meet the study objectives.
- Ensure that appropriatye laboratory QA/QC procedures are followed.

### 6.2.2 Sediment Toxicity Testing

Sediment toxicity testing is receiving increased attention as a Tier II and/or III means of assessing sediment-contaminant effects to the benthic macroinvertebrate community. It has been proposed that sediments be monitored using the Sediment Triad Approach (Chapman 1986, 1989, Long and Chapman 1985). In an AMD monitoring program this would involve using three separate investigative tools to assess sediment contamination.

1) Tier I monitoring of the concentration of contaminants in the sediments (Section IV:3.0).
2) Tier I monitoring of the benthic macroinvertebrate fauna of the sediments (Section IV:4.0).
3) Tier II monitoring investigation using sediment toxicity testing initiated with evidence of sediment-contaminant accumulation and alteration of the benthic macroinvertebrate community composition.

The American Society for Testing and Materials (ASTM) have developed standard guidelines for the collection, storage, characterization, and manipulation of sediments for laboratory toxicological testing (ASTM 1990b, 1995). The basic points are outlined below.

- Sediment collection techniques are similar to those outlined in Section 7.0 for sediment characterization.
- Sediments should be collected from both exposure stations and corresponding reference locations.
- Samples are refrigerated at 4°C, under anoxic conditions for a maximum of two weeks. Storage under anoxic conditions is important in investigations of metal toxicity for sediments collected from anoxic environments. Exposure to air may reduce or increase the potential toxicity of metals in sediments.
- Subsampling, compositing, or homogenization of sediment samples is often required for toxicity testing. Typically, the top 5 cm of the sediment is used for sediment-toxicity testing, since this is the region in which most sediment-dwelling organisms live. In order to obtain enough samples to perform the tests, sample composites may be required. If samples are composite, then they should be homogenized to distribute contaminants evenly between the test chambers.

Once sediment samples have been collected, they are transported to the laboratory to be used in sediment toxicity tests. ASTM (1995) provides standard test methods for measuring the toxicity of sediment-associated contaminants with freshwater invertebrates. The basic standard methods are provided below.

- Proper selection of test organisms.
  - -Standard methods are available for *Hyalella azteca*, *Chironomus tentans*, *C. riparius*, *C. dubia*, *Tubifex tubifex*, and *Lumbriculus variegatus*.
  - -Monitoring with organisms that do not have standard methods available should be performed cautiously, and ecological relevance should be addressed.
  - -Proper experimental design (dependent on study objectives):
    - a negative control sediment used to assess the acceptability of a test;
    - test sediment collected from the area of concern;
    - reference sediment collected from the corresponding reference station used in the monitoring program;
    - replicated test chambers for all sediments(control, test and reference); and
    - same number of test organisms in each chamber (typically 10).
- Selection of appropriate endpoints (e.g., acute tests: mortality or immobility; chronic tests: growth rate, reproductive output).
- Duration of toxicity test (10 day acute tests, several weeks for chronic tests).
- Description of sediment characteristics (particle size, metal concentrations, pH of overlying water, Eh of sediment, and other relevant information).
- Good laboratory practices (QA/QC).

### 6.3 Section Summary

The standard use of toxicity testing has been to determine in the laboratory contaminant- specific acute toxicity, to establish national and regional standards or surface water-quality guidelines. It also serves as a source-monitoring component to assess the potential biological toxicity of effluent released to the receiving environment. A third role for toxicity testing involves its use as a Tier II or III investigative tool for determining cause-and-effect relationships for biological effects detected in the receiving environment.

Toxicity tests are often classified into four types, single species tests, multi-species tests, ecosystem tests, and mesocosm studies. **Single-species** toxicity tests are the standard protocol for toxicity assessments and are typically performed in controlled laboratory settings. **Multi-species** tests involve the use of laboratory microcosms or small-scale enclosures containing samples from the natural ecosystem and incorporate several different species, typically fish, invertebrates, and plants.

**Ecosystem** tests are usually field experiments involving treatment manipulations of whole ponds/lakes or the subdivision of a single lake into several test areas. **Mesocosm** studies bridge the gap between laboratory multi-species studies and field ecosystem studies. These studies usually involve the use of multiple species established in replicated artificial ponds and/or streams and have excellent potential as Tier II or III investigative tools.

### Role in Tiered Monitoring

With the exception of single-species compliance effluent bioassays, the role of toxicity tests in monitoring is generally restricted to Tier II and/or III investigative monitoring. It is not possible to discuss all of the toxicity testing options for Tier II and III investigations, as their value will be entirely dependent on the site-specific aspects of the observed Tier I effect.

**Single-species screening toxicity tests** of effluent are Tier I compliance components used to determine toxicity prior to release into the environment. If the AMD is found to cause significant mortality to test organisms, then Tier II screening through **Toxicity Identification Evaluation** (TIE) should be performed to determine the causative agent(s).

**Sediment toxicity testing** is receiving increased attention as a Tier II and/or III means of assessing sediment contaminant effects to the benthic macroinvertebrate community. In an AMD monitoring program sediment toxicity testing is best completed as a component of a Sediment Triad study design and as such may become a Tier I component. **Mesocosm** studies are also rapidly gaining recognition as potentially excellent Tier III investigative tools for determining cause-and-effect relationships.

## 6.4 Section Bibliography

**American Standard Methods**

ASTM 1988a,b,
APHA 1992

**Artificial Streams**

Bothwell 1993
Culp and Podemski. 1995
Culp et al. 1996a
Culp et al. 1996b
Craig 1993
Gelwick and Matthews 1993
Guckert 1993
Hauer, R.F. 1993
Lamberti 1993
Lamberti and Steinman 1993
Lowell et al. 1995
McIntire 1993
Swift et al. 1993

**Sediment Toxicity Assays**

ASTM 1990b, 1993, 1995a,b
Ankley et al. 1993, 1994
Becker et al. 1995
Benoit et al. 1993
Burton 1991
Day 1995
Env. Can. 1994b, 1995
Milani et al. 1996
Reynoldson et al. 1994
Suedal 1996

**Sediment Pore Water / Elutriate Assays**

Ankley et al. 1991

**Water Quality Toxicity Tests**

Env. Can. 1992, c,d,e,f
Masters et al. 1991
Suter et al. 1987
U.S. EPA 1989c, 1993, 1994

**Statistics for Toxicology Data**

Lee et al. 1995
Newman 1995

## 7.0 LITERATURE AND BIBLIOGRAPHY REFERENCES

Ackermann, F.A. 1980. A procedure for correcting the grain size effect in heavy metal analysis of estuarine and coastal sediments. Environ. Technol. Lett., 1: 518-527.

Ackermann, F.A., H. Bergmann, and U. Schleichert. 1983. Monitoring of heavy metals in coastal and estuarine sediments - a question of grain size: <20 • m versus <60• m. Environmental Technology Letters. 4: 317-328.

Adams, D.D. 1994. Sediment pore water sampling. In. Handbook of Techniques for Aquatic Sediments Sampling, 2nd Ed. Mudroch and MacKnight, S.D. (Eds.), Lewis Publ. Chelsea Michigan.

Adams, D.D., D.A. Darby, and R.J. Young. 1980. Selected analytical techniques for characterizing the metal chemistry and geology of fine grained sediments and interstitial water. In. Contaminants and Sediments, Vol 2. Baker, R.a. (Ed.), Ann Arbor Science Publ., Ann Arbor Michigan, pp. 3-28.

Adams, M. J. 1992. Errors and detection limits. *In*. Methods of Environmental Data Analysis. C.N. Hewitt (Ed.). Elsevier Science Publishers Ltd. pp. 181-212

Adams, S.M. 1990. Status and use of biological indicators for evaluating the effects of stress on fish. American Fisheries Society Symposium 8: 1-8.

Adams, S.M. and M.G. Ryon. 1994. A comparison of health assessment approaches for evaluating the effects of contaminant related stress on fish populations. Journal of Aquatic Ecosystem Health. 3(1): 15-25.

Adams, S.M., A.M. Brown, R.W. Goede. 1993. A quantitative health assessment index for rapid evaluation of fish condition in the field. Transactions of the American Fisheries Society, 122: 63-73.

Adams, S. M., W.D. Crumby, M.S. Greeley Jr., L.R. Shugart, and C.F. Saylor. 1992. Responses of fish populations and communities to pulp mill effluents: a holistic assessment. Ecotoxicology and Environmental Safety, 24: 347-360.

Adams, S.M., K.D. Ham, M.S. Greeley; R.F. Lehew, D.E. Hinton, and C.F. Saylor. 1996. Downstream gradients in bioindicator responses: Point source contaminant effects on fish health. Canadian Journal of Fisheries and Aquatic Sciences 53(10): 2177-2187.

Anderson, D.P. 1990. Immunological indicators: effects of environmental stress on immune protection and disease outbreaks. American Fisheries Society Symposium, 8: 38-50.

Anderson, R.O. and S.J. Gutreuter. 1983. Length, weight and associated structural indices In. Fisheries Techniques, Nielson, L.A. and D.L. Johnson (Eds.). American Fisheries Society, Bethesda, Maryland.

Alikhan , M.A., G. Bagatto, and S. Zia. 1990. The crayfish as a biological indicator of aquatic contamination by heavy metals. Water Research. 24: 1069-1076.

Aljoe, W.W. and J.W. Hawkins. 1991. Hydrological characterizations and in situ neutralization of acidic mine pools in abandoned underground coal mines. In. Second International Conference on the Abatement of Acidic Drainage, Tome 1, Montreal, MEND (Ed.) pp. 69-90.

Allan, J.D. 1984. Hypothesis testing in ecological studies of aquatic insects. In. The Ecology of Aquatic Insects. V.H. Resh and D.M. Rosenberg (Eds.). Praeger Pubs., New York, pp. 484-507.

Allard, M. and G. Moreau. 1986. Leaf decomposition in an experimentally acidified stream channel. Hydrobiologia 139: 109-117.

Alldredge, R.J. 1987. Sample size for monitoring of toxic chemical sites. Environmental Monitoring and Assessment, 9: 143 - 154.

Allen, H.E., G. Fu, and B. Deng. 1993. Analysis of acid-volatile sulfide (AVS) and simultaneously extracted metals (SEM) for the estimation of potential toxicity in aquatic sediments. Environmental Toxicology and Chemistry, 12: 1441-1453.

Anderson, D.P. 1990. Immunological indicators: effects of environmental stress on immune protection and disease outbreaks. American Fisheries Society Symposium 8: 38-50.

Angermeier, P. L. and R. A. Smogor. 1995. Estimating number of species and relative abundances in stream - fish communities: effects of sampling effort and discontinuous spatial distributions. Can. J. Fish. Aquat. Sci. 52:936-949.

Ankley, G.T. 1996. Evaluation of metal/acid-volatile sulfide relationships in the prediction of metals bioaccumulation by benthic macroinvertebrates. Environmental Toxicology and Chemistry, 15 (12): 2138-2146.

Ankley, G.T., M.K. Schubauer-Berigan, and J.R. Dierkes. 1991. Predicting the toxicity of bulk sediments to aquatic organisms with aqueous test fractions: pore water vs elutriate. Environ. Toxicol Chem. 10: 1359-1366.

Ankley, G.T., D.A. Benoit, R.A. Hoke, E.N.leonard, C.W. West, G.L. Phipps, V.R. Mattson, and L.A. Anderson. 1993. Development and evaluation of test methods for benthic invertebrates and sediments: effects of flow rate and feeding on water quality and exposure conditions. Arch. Environ. Contam. Toxicol. 25: 12-19.

Ankley, G.T., D.A. Benoit, J.C. Balogh, T.B. Reynoldson, K.E. Day, and R.A. Hoke. 1994. Evaluation of potential confounding factors in sediment toxicity testswith three freswtare benthic invertebrates. Environ. Toxicol. Chem. 13: 1791-1796.

Ankley, G.T., D.M. Di Toro, D.J. Hansen, W.J. Berry. 1996a. Assessing the ecological risk of metals in sediments. Environmental Toxicology and Chemistry, 15 (12): 2053-2055.

Ankley, G.T., D.M. Di Toro, D.J. Hansen, W.J. Berry. 1996b. Technical basis and proposal for deriving sediment quality criteria for metals. Environmental Toxicology and Chemistry, 15 (12): 2056-2066.

Ankley, G.T., K. Liber, D.J. Call, T.P. Markee, T.J. Canfield, and C.G. Ingersoll. 1996c. A field investigation of the relationship between zinc and acid volatile sulfide concentrations in freshwater sediments. Journal of Aquatic Ecosystem Health. 5: 255-264.

APHA (American Public Health Association). 1992. Standard methods for the examination of water and wastewater. 18th ed. A.E. Greenberg, L.S. Clesceri, A.D. Eaton (Eds). American Public Health Association/American Water Works Association/Water Environment Federation, Washington, DC.

AQUAMIN. 1997. Assessment of the aquatic effects of mining in Canada. Final Report. Prepared for Aquamin steering Group, Prepared by Aquamin working Groups 7 and 8.

Armitage, P.D., J.H. Blackburn. 1985. Chironomidae in a Pennine stream system receiving mine drainage and organic enrichment. Hydrobiologia 121: 165-172.

Arnesan, R.T., E.R. Iversen, S.T. Källqvist, M. Laake, T. Lien. 1991. Monitoring water quality during filling of the Lokken Mine: a possible role of sulfate-reducing bacteria in metals removal. In. Second International Conference on the Abatement of Acidic Drainage, Tome 3, Montreal, MEND (Ed.) pp.201-217.

Arnold J.C. 1970. A markovian sampling policy applied to water quality monitoring of streams. Biometrics, 26: 739-747.

Asper, V.L. 1988. A review of sediment trap techniques. MTS Journal 21: 18-25.

ASTM (American Society for Testing and Materials). 1988a. Standard guide for conducting acute toxicity tests with fishes, macroinvertebrates, and amphibians. American Society for Testing and Materials, Philadelphia, P.A. ASTM E729-88a.

ASTM. 1988b. Standard guide for conducting acute toxicity tests on aqueous effluents with fishes, macroinvertebrates, and amphibians. American Society for Testing and Materials, Philadelphia, P.A. ASTM E1192-88.

ASTM. 1988c. Test method for specific gravity absoption of coarse aggregate. American Society for Testing and Materials, Philadelphia, P.A. ASTM C127-88.

ASTM. 1989a. Test method for the resistance to degredation of large-size coarse aggregate by abrasion and impact in the Los Angeles machine. American Society for Testing and Materials, Philadelphia, P.A. ASTM C535-89.

ASTM. 1989b Test method for the resistance to degredation of small size coarse aggregate by abrasion and impact in the Los Angeles machine. American Society for Testing and Materials, Philadelphia, P.A. ASTM C131-89.

ASTM. 1990a. Method for analysis of soils. American Society for Testing and Materials, Philadelphia, P.A. ASTM D422-63 (reaff. 1990).

ASTM. 1990b. Standard guide for collection, storage, characterization, and manipulation of sediments for toxicological testing, designation: American Society for Testing and Materials, Philadelphia, P.A. ASTM E 1391 - 90.

ASTM. 1990c. Test methods for laboratory determinations of water (moisture) content of soil or rock. American Society for Testing and Materials, Philadelphia, P.A. ASTM D2216-90.

ASTM. 1990d Test method for soundness of aggregates by use of sodium sulfate or magnesium sulfate. American Society for Testing and Materials, Philadelphia, P.A. ASTM C131-89.

ASTM. 1991a. Standard guide for site characterization for engineering purposes , Annual book of ASTM standards, Section 4, Construction, Volume 04.08 Soil and Rock; Dimension Stone; Geosynthetics, American Society for Testing and Materials, Philadelphia, P.A. ASTM D420.

ASTM. 1991b. Standard practices for thin-walled tube sampling of soils, Annual book of ASTM standards, Section 4, Construction, Volume 04.08 Soil and Rock; Dimension Stone; Geosynthetics, American Society for Testing and Materials, Philadelphia, P.A. ASTM D1587.

ASTM. 1991c. Standard method for penetration test and split-barrel sampling of soils. Annual book of ASTM standards, Section 4, Construction, Volume 04.08 Soil and Rock; Dimension Stone; Geosynthetics, American Society for Testing and Materials, Philadelphia, P.A. ASTM D1586.

ASTM. 1991d Test method for unit weight and voids in aggregate. American Society for Testing and Materials, Philadelphia, P.A. ASTM C29-91.

ASTM. 1992. Annual book of ASTM standards. Section 11: Water and environmental technology, volume 11.04. American Society for Testing and Materials, Philadelphia, P.A.

ASTM. 1993. Standard guide for conducting sediment toxicity tests with freshwater invertebrtaes E1383-93. *In*. Annual Book of ASTM Standards - Pesticides, Resource Recovery, Hazardous Substances and Oil Spill Responses, Waste Disposal, Biological Effects. Vol. 11.05. American Society for Testing and Materials, Philadelphia, PA. pp. 1173-1199.

ASTM. 1995a. Standard test methods for measuring the toxicity of sediment-associated contaminants with freshwater invertebrates, E1706-95. *In*. Annual Book of ASTM Standards - Pesticides, Resource Recovery, Hazardous

Substances and Oil Spill Responses, Waste Disposal, Biological Effects. Vol. 11.05. American Society for Testing and Materials, Philadelphia, PA. pp. 1204-1285.

ASTM. 1995b. Standard guide for collection, storage, characterization and manipulation of sediments for toxicology testing E1706-95. *In*. Annual Book of ASTM Standards - Pesticides, Resource Recovery, Hazardous Substances and Oil Spill Responses, Waste Disposal, Biological Effects. Vol. 11.05. American Society for Testing and Materials, Philadelphia, PA. pp. 835-855.

Auer, M.T. and N.A. Auer. 1990. Chemical suitability of substrates for walleye egg development in the Lower Fox River, Wisconsin. Trans. Amer. Fish. Soc.. 119: 871-876.

Azcue, J.M., R. Rosa, and G. Lawson. 1996. An improved dialysis sampler for the in situ collection of larger volumes of sediment pore waters. Environmental Technology. 17: 95-100.

Bagenal, T. 1978. Methods for assessment of fish production in fresh watrs 3rd Ed., International Biological Program Handbook No.3. Oxford, Blackwell scientific public.

Bajpai, A.C., I.M. Calus, and J.A. Fairley. 1992. Descriptive Statistical Techniques. *In*. Methods of Environmental Data Analysis. C.N. Hewitt (Ed.). Elsevier Science Publishers Ltd.

Balasundaram and Shashidhara. 1993. Data validation practices and risk assessment. Civil Engineering, March 1993, pp. 60-61.

Bates, R.G. 1981. The modern meaning of pH. CRC Crit. Rev. Anal. Chem. 10: 247-.

Baudo, R. 1990. Sediment sampling, mapping and data analysis, In. Sediments: Chemistry and Toxicity of In-Place Pollutants. Baudo, R. J. Giesy, and H. Muntau (Eds.), Lewis Publ., Ann Arbor Michigan. pp. 15-60.

Baumann, P.C., I.R. Smith, and C.D. Metcalfe. 1996. Linkages between chemical contaminants and tumors in benthic Great Lakes fish. 1996. J. Great Lakes Res. 22(2): 131-152.

BC AMD (British Columbia Acid Mine Drainage). 1989. Draft acid rock drainage technical guide volume I. Prepared by S. Robertson, Kirsten (B.C.) Inc., Norecol Environmental Consultants, Gormely Process Engineering. BiTech Publishers Ltd. Vancouver, Canada.

BC AMD. 1990. Draft acid rock drainage technical guide volume II: summary guide.. Prepared by S. Robertson, Kirsten (B.C.) Inc., Norecol Environmental Consultants, Gormely Process Engineering. BiTech Publishers Ltd. Vancouver, Canada.

BC AMD. 1990a. The effect of treated acid mine drainage on stream macroinvertebrates and periphytic algae: An in situ mesocosm experiment. Prepared by, Limnotek Research and Development Inc., BiTech Publishers Ltd. Vancouver, Canada.

BC AMD. 1990b. Monitoring acid mine drainage. Prepared by, E. Robertson, S. Robertson, Kirsten (B.C.) Inc., BiTech Publishers Ltd. Vancouver, Canada.

BC AMD. 1990c. Review of sediment monitoring techniques. British Columbia Acid Mine Drainage Task Force, Victoria, BC., BiTech Publishers Ltd. Vancouver, Canada.

BC AMD. 1991a. Evaluation of ARD from Britannia Mine and the options for long term remediation of the impact on Howe Sound. Ministry of Mines and Petroleum Resources., BiTech Publishers Ltd. Vancouver, Canada.

BC AMD. 1991b. A preliminary biological and geological assessment of subaqueous tailings disposal in Benson Lake, British Columbia. Prepared by Rescan Environmental Services. British Columbia Acid Mine Drainage Task Force, Victoria, BC., BiTech Publishers Ltd. Vancouver, Canada.

BC AMD. 1992. Field QA/QC protocols for monitoring and predicting impacts of acid rock drainage. Prepared by Norecol Environmental Consultants for British Columbia Acid Mine Drainage Task Force, Victoria, BC., BiTech Publishers Ltd. Vancouver, Canada.

BC AMD. 1992. Stream community responses to a gradient of acid mine drainage additions. Prepared by, Limnotek Research and Development Inc., BiTech Publishers Ltd. Vancouver, Canada.

BC AMD. 1992. Use of paleolimnological techniques to assess the effects of acid rock drainage. Prepared by, Norecol Environmental Consultants Ltd., BiTech Publishers Ltd. Vancouver, Canada.

BC AMD. 1993. Literature review report: possible means of evaluating the biological effects of sub-aqueous disposal of mine taiulings. Prepared by P.G.C. Campbell and R. Roy, INRS-EAU., BiTech Publishers Ltd. Vancouver, Canada.

BC Env. 1995. Water quality assessment and objectives for Tsolum River watershed. Prepared by Water Quality Branch. Ministry of Environment Lands and Parks, Victoria, BC.

BC MELP. 1993. Revision of pollution control criteria for mining, smelting and related industries of British Columbia. Prepared by EVS Consultants and Steffen Robertson, Kirsten for British Columbia Ministry of Environment Lands and Parks, Victoria, BC.

BC MEMPR and BC MELP. 1992. A guide to the review and certification of mine developments. British Columbia Energy Mines and Petroleum Resources and Ministry of Environment, Lands and Parks, Victoria, BC.

BC MOE. 1974. Pollution control objectives for the mining, mine-milling and smelting industries of British Columbia. B.C. Ministry of Environment, Victoria, BC.

Becker, C.D., D.A. Neitzel. 1992. Water quality in North American river systems. Battelle Press, Columbia, OH.

Becker, D.S., C.D. Rose, and G.N. Bigham. 1995. Comparison of the 10-day freshwater sediment toxicity tests using *Hyallella azteca* and *Chironomus tentans*. Environ. Toxicol. Chem. 14: 2089-2094.

Beland, K. F., R. M. Jordan, and A. L. Meister. 1982. Water depth and velocity preferences of spawning Atlantic salmon in Maine rivers. N. Amer. J. Fish. Manage. 2:11_13.

Bendell-Young, L.I., H.H. Harvey, J.F. Young. 1986. Accumulation of cadmium by white suckers (Catostomus commersoni) in relation to fish growth and lake acidification. Can. J. Fish. Aquat. Sci., 43: 806-811.

Benoit, D.A., G.L. Phipps, and G.T. Ankley. 1993. A sediment testing intermittent renewal system for the automated renewal of overlying water in toxicity tests with contaminated sediments. Water Res. 27(9): 1403-1412.

Bennett, J.W. and G. Pantelis 1991. Construction of a waste rock dump to minimize acid mine drainage: a case study. In. Second International Conference on the Abatement of Acidic Drainage, Tome 3, Montreal, MEND (Ed.) pp. 299-318.

Berner, R. 1981. A new geochemical classificatio of sedimentary environment, Journal of Sedimentary Petrology, 51: 359-365.

Bernstein, B.B., J. Zalinski. 1983. An optimum sampling design and power tests for environmental biologists. Journal of Environmental Management 16: 35-43.

Berry, W.J., D.J. Hansen, J.D. Mahony, D.L. Robson, D.M. Di Toro, B.P. Shipley, B. Rogers, J.M. Corbin, W.S. Boothman. 1996. Predicting the toxicity of metal-spiked laboratory sediments using acid-volatile sulfide and interstitial water normalizations. Environmental Toxicology and Chemistry, 15 (12): 2067-2079.

Besser, J.M., C.F. Rabeni. 1987. Bioavailability and toxicity of metals leached from lead-mine tailings to aquatic invertebrates. Environmental Toxicology and Chemistry, 6: 879-890.

Besser, J.M., Giesy, J.P., Brown, R.W., and G.A. Dawson. 1995. Bioaccumulation of selenium and associated environmental hazards in a freshwater fish community.

Besser, J.M., C.G. Ingersoll, J.P. Giesy. 1996. Effects of spatial and temporal variation of acid-volatile sulfide on the bioavailability of copper and zinc in freshwater sediments. Environmental Toxicology and Chemistry, 15 (3): 286-293.

Bjornn, T. C., and D. W. Reiser. 1991. Habitat requirements of salmonids in streams. A. F. S. Special Publ. 19:83_138.

Black, J.J. 1984. Aquatic animal neoplasia as an indicator for carconogenic hazards to man. Hazard Assessment of Chemicals, 3: 181-232

Blazer, V.S., R.E. Wolke, J. Brown, C.A. Powell. 1987. Piscine macrophage aggregate parameters as health monitors; effect of age, sex, relative weight, season and site quality in largemouth bass (Micropterus salmoides) Aquatic Toxicology, 10: 199-215.

Bodo, B.A. 1991. Trend analysis and mass-discharge estimation of atrazine in southwestern Ontario Great lakes tributaries. Environmental Toxicology and Chemistry, 10: 1105-1121.

Bolger, T., P.L. Connolly. 1989. The selection of suitable indices for the measurement and analysis of fish condition. J. Fish Biol., 34: 171-182.

Bond, W.R. and J.V. Rouse. 1985. Lysimeters allow quicker monitoring of heap leaching and tailing site. Mining Engineering, 37: 314-319.

Bond, W.R. 1995. Case studies of vadose zone monitorin and sampling using porous suction cup samplers. In. Handbook of Vadose Zone Characterization and Monitoring. Wilson, L.G., L.G. Everett, and S.J. Cullen. Lewis Publ. Ann Arbor Michigan. pp. 523-532.

Booty, W.G., D.C.L. Lam, I. Wong, G.S. Bowen, D.A. Swayne. Water quality modelling system for acid mine drainage. In. Second International Conference on the Abatement of Acidic Drainage, Tome 2, Montreal, MEND (Ed.) pp. 427-447.

Bowell, R.J., A. Warren, H.A. Minjera, and N. Kimaro. 1995. Environmental impact of former gold mining on the Orangi River, Serengeti N.P., Tanzania. Biogeochemistry, 28(3): 131-160.

Bothwell, M.L. 1993. Artificial streams in the study of algal/nutrient dynamics. Journal of the North American Benthological Society, 12 (4): 327-333.

Bowen, G.S., D.C.L. Lam, I. Wong, D. Swayne, and J. Storey. 1991. A microcomputer-based system for data management and impact assessment of mine effluents. In. Second International Conference on the Abatement of Acidic Drainage, Tome 2, Montreal, MEND (Ed.) pp. 271-289.

Bovee, K.D. 1978. The incremental method of assessing habitat potential for coolwater species, with management implications. Am. Fish. Soc. Spec. Publ. 11:340-346.

Bowen, G.S., D.C.L. Lam, I. Wong, D. Swayne, J. Storey. 1991. A microcumputer-based system for data management and impact assessment of mine effluents. In. Second International Conference on the Abatement of Acidic Drainage, Tome 2, Montreal, MEND (Ed.) pp. 272-289.

Brassard, P, F. Rosa, and A. Mudroch. 1996. Disposal of acid drainage generating mine tailings in lakes and man-made reservoirs. Environmental Technology 17(10): 1059-1069.

Brassington, R. 1990. Field hydrology. Geological Society of London Professional Handbook Series, Open University Press, Milton Keynes & Halsted Press, John Wiley & Sons, New York - Toronto.

Brix, H. and J.E. Lyngby 1983. te distribution of some metallic elements in eel grass (*Zostera marina*) and sediment in the Limnfjord , Denmark. Estuarine, Coastal and Shelf Science 16: 19-22.

Brothers, E.B. 1987. methodological approaches to the examination of otoliths in aging studies. *In*. Age and Growth of Fish. R.C. Summerfelt and G.E. Itall (Eds.). Iowa State Univ. Press. Ames.

Brouwer, H. And T.P. Murphy. 1994. Diffusion method for the determination of acid-volatile sulphides (AVS) in sediment. Environmental Toxicology and Chemistry 13(8): 1273-1275.

Brooke, L.T., G.T. Ankley, D.J. Call, P.M. Cook. 1996. Gut content weight and clearance rate for three species of freshwater invertebrates. Environmental Toxicology and Chemistry, 15 (2): 223-228.

Broshears, R.E. 1996. Reactive solute transport in acidic streams. Water, Air and Soil Pollution, 90(1-2): 195-205.

Brumbaugh, W.G., C.G. Ingersoll, N.E. Kemble, T.W. May, J.L. Zajicek. 1994. Chemical characterization of sediments and pore water from the upper Clark Fork River and Milltown Reservoir, Montana. Environmetnal Toxicology and Chemistry, 13(12): 1971-1983.

Brumbaugh, W.G., J.W. Arms. 1996. Quality control considerations for the determination of acid-volatile sulfide and simultaneously extracted metals in sediments. Environmental Toxicology and Chemistry, 15 (3): 282-285.

Buckley, R. 1991. Auditing the precision and accuracy of environmental impact predictions in Australia. Environmental Monitoring and Assessment, 18: 1-23.

Bufflap, S.E. and H.E. Allen. 1995a. Sediment pore water collection methods for trace metal analysis: a review. Wat. Res. 29(1): 165-177.

Bufflap, S.E. and H.E. Allen. 1995b. Comparison of pore water sampling techniques for trace metals. Wat. Res. 29(9): 2051-2054.

Bumb, A.C., C.R. McKee, R.B. Evans, and L.A. Eccles. 1988. Design of lysimeter leak detector networks for surface impoundments and landfill. Ground Water Monitoring Review Journal, 8: 102-114.

Burd, B.J., A. Nemec, R.O. Brinkhurst. 1990. The development and application of analytical methods in benthic marine infaunal studies. Adv. Mar. Biol. 26: 169-247.

Burton, Jr., A.G. 1991. Assessing the toxicity of freshwater sediments. Environmental Toxicology and Chemistry, 10: 1585-1627.

Busch, W.D.N and P.G. Sly (eds.). 1992. The development of an aquatic habitat classification system for lakes. CRC Press, Boca Raton, FL.

Byerly, D.W. 1990. Guidelines for handling ecavated acid-producing material. In. Designing for closure, J.W. Gadsby, J.A. Malick, and S.J. Day. (Eds.), BiTech Publ. Vancouver, B.C., pp 109-129.

Cailas, M.D., G. Cavadias, and R. Gehr. 1986. Application of a nonparametric approach for monitoring and detecting trends in water quality data of the St. Lawrence River. Water Pollut. Res. J. Canada, 21: 153-167.

Cameco 1994. Environmental impact statement: Deilmann in-pit tailings management facility. Prepared by Cameco Environment and Safety Dept. , Prepared for Mineral Industry environmental Protection Branch, Atomic Energy Control Board.

Carignan, R., F. Rapin, and A. Tessier. 1985. Sediment porewater sampling for metal analysis: a comparison of techniques. Geochem. Cosmochim. Acta. 49: 2493-2497.

Carigan R., S. St-Pierre, and R. Gächter. 1994. Use of diffusion samplers in oligotrophic lake sediments: effects of free oxygen in sampler material. Limnol. Oceanogr. 39(2): 468-474.

Carins, Jr., J., and P.V. McCormick. 1992. Developing an ecosystem-based capability for ecological risk assessments. The Environmental Professional, 14: 186-196.

Carlson, A.R., G.L. Phipps, V.R. Mattson, P.A. Kosian, and A.M. Cotter. 1991. The role of acid-volatile sulfide in determining cadmium bioavailability and toxicity in freshwater sediments. Environmental Toxicology and Chemistry, 10: 1309-1319.

Carter, M.R. (ed.) 1993. Soil sampling and methods of analysis. Lewis Publishers, London, U.K.

Casey, R.J. 1994. Artificial substrata as quantitative sampling devices of benthic macroinvertebrates in flowing water habitats. Alberta Environmental Centre, Vegreville, Alberta.

Casselman, J.M. 1987. Determination of age and growth. *In*. The Biology of Fish Growth. H. Weatherly and H.S. Gillo (Eds.). Academic Press London.

Casselman, J. M. and C. A. Lewis. 1996. Habitat requirements of northern pike (*Esox lucius*). Can. J. Fish. Aquat. Sci. 53(Suppl. 1):161-174.

CCME. 1991a. A protocol for developing the derivation of water quality guidelines for the protection of aquatic life. In. CCCREM 1987. Canadian Water Quality Guidelines, Appendix IX. Winnipeg, Manitoba. 8pp

CCME. 1991b. Interim Canadian environmental quality criteria for contaminated sites. Report CCME EPC-CS34. Prepared by the CCME Subcomittee on environmental quality criteria for contaminated sites.

CCME. 1992. National classification system for contaminated sites. Report CCME EPC-CS39E. CCME subcommittee on Classification of Contaminated Sites. Winnipeg, Manitoba, pp. 54.

CCME 1995. Protocol for the derivation of Canadian sediment quality guidelines for the protection of aquatic life. Report CCME EPC-98E. Prepared by the task group on water quality guidelines, Winnipeg, Manitoba.

CCME. 1996. Canadian water quality guidelines. Task Force on Water Quality Guidelines of the Canadian Council of Resources and Ministers of the Environment. Updated to May 1996. Ottawa, ON.

Cedergren, H.R. 1989. Seepage, drainage and flow nets. Wiley-Interscience. John Wiley & Sons, New York - Toronto.

Chadwick, J.W., and S.P. Canton. 1984. Inadequacy of diversity indices in discerning metal mine drainage effects on a stream invertebrate community. Water, Air and soil Pollution, 22(2): 217-223.

Chadwick, J.W., S.P. Canton, and R.L. Dent 1986. Recovery of benthic invertebrate communities in Silver Bow Creek, Montana, following improved metal mine wastewater treatment. Water, Air and Soil Pollution, 28(3/4): 427-438.

Chao, T. 1984. Use of partial dissolution techniques in geochemical exploration. Journal of Geochemical Exploration. 20: 101-106.

Chapman, P.M. 1986. Sediment quality criteria from the sediment quality triad: An example. Environmental Toxicology and Chemistry, 5: 957-964.

Chapman, P.M. 1989. Current approaches to developing sediment quality criteria. Environmental Toxicology and Chemistry, 8: 589-599.

Chapman, P.M. 1996. SEM:AVS does not predict sediment toxicity. SETAC News, 16 (2): 13-14.

Chapman, P.M., G. P. Romberg, and G.A. Vigers. 1982. Design of monitoring studies for priority pollutants. Journal WPCF, 54 (3): 292-297.

Chapman, P.M., R.C. Barrick, J.M. Neff, and R.C. Swartz. 1987. Four independent approaches to developing sediment quality criteria yield similar values for model contaminants. Environmental Toxicology and Chemistry, 6: 723-725.

Chapman, P.M., E.R. Long, R.C. Swartz, T.H. DeWitt, and R. Pastorok. 1991. Sediment toxicity tests, sediment chemistry and benthic ecology do provide new insights into the significance and management of contaminated sediments - A reply to Robert Spies. Environmental Toxicology and Chemistry, 10: 1-4.

Chapman, P.M., E.A. Power, and G.A. Burton. 1992. Integrative assessments in aquatic ecosystems. In. Sediment Toxicity Assessment. Burton, G.A. (Ed.), Lewis Publ. Ann Arbor Michigan. pp. 313-340.

Chapman, P.M., R.S. Caldwell, and P.F. Chapman. 1996. Letter to the editor: A warning: NOECs are inappropriate for regulatory use. Environmental Toxicology and Chemistry, 15 (2): 77-79.

Chigbu, P and T.H. Sibly. 1994. Relationship between abundance, growth, egg size and fecundity in a landlocked population of longfin smelt, Spirinchus thaleichthys. Journal of Fish Biology. 45(1): 1-15.

Clarke, K.R. 1993. Non-parametric community analyses of changes in community structure. Australian Journal of Ecology, 18(1): 117-143

Clements, W.H. 1994. Benthic Invertebrate Community Responses to Heavy Metals in the Upper Arkansas River Basin, Colorado. Journal North American Benthological Soc. 13(1): 13-44.

Clements, W.H., P.M. Kiffney. 1994. Assessing contaminant effects at higher levels of biological organization. Environmental Toxicology and Chemistry, 13 (3): 359-359.

Clements, W.H., D.S. Cherry and J. Cairns, Jr. 1988. Impact of heavy metals on insect communities in streams: a comparison of observational and experimental results. Can. J. Fish. Aquat. Sci. 45: 2017-2025.

Clements, W.H., D.S. Cherry and J.H. Van Hassel. 1992. Assessment of the impact of heavy metals on benthic communities at the Clinch River (Virginia): evaluation of an index of community sensitivity. Can. J. Fish. Aquat. Sci. 49: 1686-1694.

Clifford, P.A., Barchers, D.E., Ludwig, D.F., Sielken, and R.L. Klingensmith. 1995. Hazard/risk assessment: an approach to quantifying spatial components of exposure for ecological risk assessment. Environmental Toxicology and Chemistry, 14(5): 895-906.

Coggans, C.J., D.W. Blowes and W.D. Robertson. 1991. The hydrogeology and geochemistry of a nickel-mine tailings impoundment, Copper Cliff, Ontario. In. Second International Conference on the Abatement of Acidic Drainage, Tome 4, Montreal, MEND (Ed.) pp. 126.

Coker, W.B. and C.E. Dunn 1983. Lake water and lake sediment geochemistry, NEA/IAEA Athabasca test Area. In. Uranium Exploration in Athabasca Basin, Saskatchewan, Canada. Cameron, E.M. (Ed.). Geological Survey of Canada. pp. 117-125.

Collings, M. R. 1974. Generalization of spawning and rearing discharges fro several Pacific salmon species in western Washington. U. S. Geol. Surv. Open File Report, Tacoma, Washington.

Conover, W.J. 1980. Practical nonparametric statistics. 2nd ed. John Wiley & Sons, New York, NY.

Cook, M. F. and E. P. Bergersen. 1988. Movements, habitat selection, and activity periods of northern pike in Eleven Mile Reservoir, Colorado. Trans. Amer. Fish. Soc. 117:495_502.

Cook, M. F., and R. C. Solomon. 1987. Habitat suitability index models: muskellunge. Biological Report 82(10.148

Cope, G.W., J.G. Wiener, G.J. Atchison. 1994. Hepatic cadmium metal-binding proteins and bioaccumulation in bluegills exposed to aqueous cadmium. Environmental Toxicology and Chemistry, 13 (4): 553-562.

Corbett, B. W. and P. M. Powles. 1986. Spawning and larvae drift of sympatric walleyes and white suckers in an Ontario stream. Trans. Amer. Fish. Soc. 115:41_46.

Cornwell, J.C. 1985. Sediment accumulation rates in an Alaskan Arctic lake using a modified $^{210}$Pb Technique. Can. J. Fish. Aquat. Sci., 42: 809-814.

Cowardin, L.M., V. Carter, F.C. Golet, and E.T. LaRoe. 1979. Classification of wetlands and deepwater habitats of the United States. U.S. Fish and Wildlife Service, FWS/OBS-79/31.

Craig, J.F., A. Sharma, and K. Smiley. 1986. The variability in catches from multi-mesh gillnets fished in three Canadian lakes. J. Fish. Biol. 28: 671-678.

Craig, J.F., J.A. Babaluk. 1989. Relationship of condition of walleye (Stizostedion vitreum) and northern pike (Esox lucius) to water clarity, with special reference to Dauphin Lake, Manitoba. Can. J. Fish Aquat. Sci., 46: 1581-1586.

Craig, D.A. 1993. Hydrodynamic considerations in artificial stream research. Journal of the North American Benthological Society, 12 (4): 324-327.

Cranston, P.S. 1990. Biomonitoring and invertebrate taxonomy. Environmental Monitoring and Assessment. 14: 265-273.

Crawford, J.K. and S.N. Luoma. 1993. Guideline for studies of contaminants in biological tissues for the national water-quality assessment program. U.S. Geological Survey Open file report 92-494, lemoyne, Pennsylvania.

Crossey, M.J. and T.W. LaPoint. 1988. Comparison of periphyton community structural and functional responses to heavy metals. Hydrobiologia, 162(2): 109-121.

Cuffney, T.F., M.E. Gurtz, and M.R. Meador. 1993. Guidelines for the processing and quality assurance of benthic invertebrate samples collected as part of the national water-quality assessment program. U.S. Geological Survey, Open- file report 93-406.

Culp J.M. and C.L. Podeski. 1995. Design and application of a novel stream microcosm system for assessing effluent impacts to large rivers. In. fate and Effects of Pulp and Paper Mill Effluents. M.R. Servos, K.R. Munkittrick, J.H. Carey, and G.J. Van Ser Kraak (Eds.). St. Lucie Press, Boca Raton, Florida.

Culp, J.M., C.L. Podemski, C. Casey. 1996a. Design and application of a transportable experimental stream system for assessing effluent impacts on riverine biota. Northern River Basins Study.

Culp, J.M., C.L. Podemski, K.J. Cash, R.B. Lowell. 1996b. Utility of field-based artificial streams for assessing effluent effects on riverine ecosystems. Journal of Aquatic Ecosystem Health, 5: 117-124.

Cushing, C.E. and J.M. Thomas. 1980. 1980. Cu and Zn kinetics in Myriophyllum heterophyllym Michz. and Potamogeton richardsonii (Ar. Benn) Rydbd. Ecology 61: 1321-1326.

Czarnezki, J.M. 1987. Use of the pocketbook mussel, *Lampsilis ventricosa*, for monitoring heavy metal pollution in an Ozark stream. Bulletin of Environmental Contamination and Toxicology. 38: 641-646.

Davies W.D. 1983. Sampling with toxicants. In. Fisheries Techniques L. A. Neilsen and D. L. Johnson (Eds.). American Fisheries Society. Bethesda, Maryland. pp. 199-214.

Davison, A.C. 1992. Regression and Correlation. In: Methods of Environmental Data Analysis, (ed) C.N. Hewitt. Elsevier Science Publishers Ltd. pp. 79-137.

Davison, W., H. Zhang, and G.W. Grime. 1994. Performance characteristics of gel probes used for measuring the chemistry of pore waters. Environ. Sci. Technol. 28: 1623-1632.

Day, K.E., R.S. Kirby, and T.B. Reynoldson. 1995. The effect of manipulations of freshwater sediments on responses of benthic invertebrates in whole-sediment toxicity tests. Environ. Toxicol. Chem. 14: 1333-1343.

Day, S., G. Hope, W. Kuit. 1977. Waste rock management planning for the Kudz Ze Kayah Project, Yukon Territory 1: Predictive static and kinetic test work. Proceedings of the 4th International Conference on Acid Rock Drainage. Vol. 1: 81-98.

Day, R.W. and G.P. Quinn. 1989. Comparisons of treatments after an analysis of variance in ecology. Ecol. Monog., 59: 433-463.

Day, K., T.B. Reynoldson. 1995. Ecotoxicology of depositional sediments, Athabasca River, May and September 1993. Northern River Basins Study.

Demayo, A., S. Whitlow. 1986. Graphical presentation of water quality data. IAHS Publ. No. 157.

de Groot, A.J., K.H. Zschuppe, and W. Salomons. 1982. Standardization of methods of analysis for heavy metals in sediments. Hydrobiologia. 92: 689-695.

DeWitt, T.H., R.C. Swartz, D.J. Hansen, D. Mcgovern, W.J. Berry. 1996. Bioavailability and chronic toxicity of cadmium in sediment to the estuarine amphipod Leptocheirus plumulosus. Environmental Toxicology and Chemistry, 15 (12): 2095-2101.

Dermott, R.M. 1985. Benthic fauna in a series of lakes displaying a gradient of pH. Hydrobiologia, 128: 31-38.

DFO (Department of Fisheries and Oceans ) and B.C. MEP 1987. Fish habitat inventory and information program - stream survey field guide.

DFO and Env. Can. 1992. Annex 1: Aquatic environmental effects monitoring requirements at pulp and paper mills and off-site treatment facilities regulated under the pulp and paper effluent regulations of the fisheries act.

DFO and Env. Can. 1995a. Further guidance for the adult fish survey for aquatic environmental effects monitoring related to federal Fisheries Act Requirements. EEM 1. Dept. of Fisheries and Oceans Env. Canada, Ottawa.

DFO and Env. Can. 1995b. Further guidance for the invertebrate community survey for aquatic environmental effects monitoring related to federal Fisheries Act Requirements. EEM 2. Dept. of Fisheries and Oceans Env. Canada, Ottawa.

Diana, J. S., W. C. Mackay, and M. Ehrman. 1977. Movements and habitat preference of northern pike, *Esox lucius*, in Lac Ste. Anne, Alberta. Trans. Amer. Fish. Soc. 106:560_565.

Dickerson, K.K., W.A. Hubert, H.L. Bergman. 1996. Toxicity assessment of water from lake and wetlands receiving irrigation drain water. Environmental Toxicology and Chemistry, 15 (7): 1097-1101.

Diks, D.M. and H.E. Allen. 1983. Correlation of copper distribution in a freshwater-sediment system to bioavailability. Bull. Environ. Contam. Toxicol., 30: 37-43.

Di Toro, D.M., J.D. Mahony, D.J. Hansen, K.J. Scott, A.R. Carlson, G.T. Ankley. 1992. Acid volatile sulfide predicts the acute toxicity of cadmium and nickel in sediments. Environ. Sci. Technol., 26: 96-101.

Di Toro, D.M., J.D. Mahony, A.M. Gonzalez. 1996a. Particle oxidation model of iron and cadmium sulfide in sediments. Environmental Toxicology and Chemistry, 15 (12): 2156-2167.

Di Toro, D.M., J.D. Mahony, D.J. Hansen, W.J. Berry. 1996b. A model of the oxidation of iron and cadmium sulfide in sediments. Environmental Toxicology and Chemistry, 15 (12): 2168-2186.

Domenico, P.A. and F.W. Schwartz. 1990. Physical and chemical hydrogeology. John Wiley and Sons.

Donaldson, E.M. 1990. Reproductive indices as measures of the efffets of environmental stress on fish. American Fisheries Society Symposium 8: 109-122.

Downing, J.A. 1984. Sampling the benthos of standing waters. In. A Manual on Methods for the Assessment of Secondary Productivity in Fresh Waters, 2nd Ed. Downing, J.A. and F.H. Rigler (Eds.). IBP Handbook 17. Blackwell Scientific Pubs. pp. 87-130.

Doyon, J.F., J.A. Downing, and E. Magnin. 1988. Variation in the condition of northern pike, (Esox lucius). Can. J. Fish. Aquat. Sci., 45: 479-482.

Draper, N. and R., H. Smith. 1981. Applied regression analysis. 2nd ed. John Wiley & Sons, New York, NY.

Driscoll, F. 1989. Groundwater and wells. 2nd ed. Johnson Filtration Systems, Inc, St. Paul, MN., H.M. Smyth Co. Inc.

Duncan, D. W. And A. Bruynesteyn. 1979. Determination of the acid production potential of waste materials. AIME Annual Meeting, New Orleans, Louisiana, U.S.A. Paper A-79-29.

Dunne,T. and L.B. Leopold 1978. Water in environmental planning. W.H. Freeman and Co.

Dunnicliff, J., G.E. Green. 1988. Geotechnical instrumentation for monitoring field performance. John Wiley & Sons, New York, NY.

Dutton, M.D., M. Stephenson, J.F. Klaverkamp. 1993. A mercury saturation assay for measuring metallothionein in fish. Environmental Toxicology and Chemistry, 12: 1193-1202.

Eberhardt, L.L. and J.M. Thomas. 1991. Designing environmental field studies. Ecological Monographs, 6 (1): 53-73.

Eco/Log. 1992. Metal mining liquid effluent guidelines. Can. Pollut. Legislation, 6: 283-285.

Edward E. Johnson Inc. 1959. The principles and practical methods of developing water wells. Edward E. Johnson Inc., St. Paul, MN.

Edwards, E. A. 1983a. Habitat suitability index models: bigmouth buffalo. FWS/OBS 82/10.34.

Edwards, E. A. 1983b. Habitat suitability index models: longnose dace. FWS/OBS 82/10.33.

Edwards, E. A. 1983c. Habitat suitability index models: longnose sucker. FWS/OBS 82/10.35.

Edwards, E. A., G. Gebhart, and O. E. Maughan. 1983. Habitat suitability index models: smallmouth bass. FWS/OBS 82/10.36.

Electric Power Research Institute. 1985. Field measurement methods for hydrogeologic investigations: a critical review of the literature. EPRI EA-4301 Project 2485-7, palo Alto, CA.

Env. Can. 1974. Water quality data; British Columbia, 1961-1971. Environment Canada, Ottawa, ON.

Env. Can. 1983. Sampling for water quality. Water Quality Branch, Inland Water Directorate, Ottawa, ON.

Env. Can. 1985. Sampling and analysis in the arctic marine benthic environment: vol I - review of methods. Environmental Protection Service, Ottawa, Ontario.

Env. Can. 1992a. Status report on water pollution. Control in the Canadian metal mining industry (1990 and 1991). Report EPS 1/MM/4. Environment Canada, Ottawa, ON.

Env. Can. 1992b. Aquatic environmental effects monitoring requirements. Environment Canada and Department of Fisheries and Oceans, Ottawa, ON.

Env. Can. 1992c. Biological test method: Growth inhibition test using the *Selenastrum capricornutum*. Report EPS 1/RM/25. Environment Canada Conservation and Protection Service, Ottawa, Ont.

Env. Can. 1992d. Biological test method: test of larval growth and survival using fathead minnows. Report EPS 1/RM/22. Environment Canada Conservation and Protection Service, Ottawa, Ont.

Env. Can. 1992e. Biological test method: test of reproduction and survival using the Cladoceran Ceriodaphnia dubia. Report EPS 1/RM/21. Environment Canada Conservation and Protection Service, Ottawa, Ont.

Env. Can. 1992f. Toxicity tests using early life stages of Salmonid fish (Rainbow trout, Coho Salmon, or Atlantic Salmon). Report EPS 1/RM/22. Environment Canada Conservation and Protection Service, Ottawa, Ont.

Env. Can. 1993a. Technical guidance manual for aquatic environmental effects monitoring related to federal fisheries act requirements, version 1.0. Environment Canada, Ottawa, ON.

Env. Can. 1993b. Guidelines for monitoring benthos in freshwater environments. Prepared by EVS Consultants for Environment Canada, Ottawa, Ontario.

Env. Can. 1993c. Quality assurance in water quality monitoring. Ecosystem Sciences and Evaluation Directorate, Conservation and Protection, Ottawa, Ontario.

Env. Can. 1994a. Guidance document on collection and preparation of sediments for physicochemical characterization and biological testing. Environmental Protection Series report EPS 1/RM/30. Ottawa. Ontario.

Env. Can. 1994b. Guidance document on measurement of test precision using control sediments spiked with a reference toxicant. Environmental Protection Series report EPS 1/RM/29. Technology Development Directorate, Ottawa, Ontario.

Env. Can. and DFO. 1991. Technical guidance manual for aquatic environmental effects monitoring at pulp and paper mills volume I: Overview and study design.

Env. Can. and DFO. 1992. Technical guidance document for aquatic environmental effects monitoring, draft.

Env. Can. and DFO. 1993. Technical guidance document for aquatic environmental effects monitoring related to federal fisheries act requirements, version 1.0.

Env. Can. and NHRI. 1992. Natural acid rock drainage at Macmillan Pass, Yukon. Prepared by, Y.T. J. Kwong, W.G. Whitley.

Erickson, R.J., D.A. Benoit, V.R. Mattson, H.P. Nelson Jr., E.N. Leonard. 1996. The effects of water chemistry on the toxicity of copper to fathead minnows. Environmental Toxicology and Chemistry, 15 (2): 181-193.

Errington, J.C., K.D. Ferguson. 1987. Acid mine drainage in British Columbia - today and tomorrow. In: Proceedings of the Acid Mine Drainage Seminar/Workshop, Halifax, NS, 23-26 March 1987. Ministry of Supply and Services, pp. 67-88.

Errington, J.C. 1991. The regulation of acid mine drainage. In. Second International Conference on the Abatement of Acidic Drainage, Tome 2, Montreal, MEND (Ed.) pp. 89-100.

Evaldi, R.D. and R. Garcia. 1991. Quality of South Fork Cumberland River near Stearns, Kentucky. In. Second International Conference on the Abatement of Acidic Drainage, Tome 3, Montreal, MEND (Ed.) pp. 417-434.

Faith, D.P., C.L. Humphrey, and P.L. Dostine. 1991. Statistical poer and BACI designs in biological monitoring: caomparative evaluation of measures of community dissimilarity based on benthic macroinvertebrate communities in Rockhole Mine Creek, Northern territory, Australia. Aust. J. Mar. Freshwater res. 42: 589-602.

Fairchild, J.F., T. Boyle, W.R. English, C. Rabeni. 1987. Effects of sediment and contaminated sediment on structural and functional components of experimental stream ecosystems. Water, Air, and Soil Pollution, 36: 271-293.

Fausch, K.D., J.R. Karr, and P.R. Yant. 1984. Regional application of an index of biotic integrity based on stream fish communities. Transactions of the American Fisheries Society, 113: 39-55.

Fausch, K. D., C. L. Hawkes and M. G. Parsons. 1988. Models that predict standing crop of stream fish from habitat variables: 1950-1985. Gen. Tech. Rep. PNW-GTR-213. Portland, Oregon. U. S. Dept. of Agric., Forest Service. Pacific Northwest Research Station..

Fausch, K.D., J. Lyons, J.R. Karr, and P.L. Angermeier. 1990. Fish communities as indicators of environmental degredation. American Fisheries Society Symposium 8: 123-144.

Farag, A.M., M.A. Stansbury, C. Hogstad, E. MacConnell, H.L. Bergman. 1995. The physiological impairment of free-ranging brown trout exposed to metals in the Clark Fork River, Montana. Can. J. Fish Aquat. Sci., 52: 2038-2050.

Feasby, G., G. Tremblay, and M. Blanchette. 1991. The mine environment neutral drainage (MEND) program. In. Second International Conference on the Abatement of Acidic Drainage, Tome 1, Montreal, MEND (Ed.) pp. 1-26.

Ferraro, S.P., F.A. Cole, W.A. DeBen, R.C. Swartz. 1989. Power-cost efficiency of eight macrobenthic sampling schemes in Puget Sound, Washington, USA. Can. J. Fish. Aquat. Sci., 46: 2157-2165.

Filipek, L. and R. Owen. 1979. Geochemical associationas and grain size partitioning of heavy metals in lacustrine sediments. Chemical Geology, 26: 105-117.

Finkelman, R. and D. Giffin. 1986. Hydrogen peroxide oxidation: an improved method for rapidly assessing acid-generating potential of sediments and sedimentary rocks. Reclamation and Revegetation Research, 5:521-534.

Fisheries and Environment Canada. 1977. Fisheries act, metal mining liquid effluent regulations and guidelines (MMLER). Environmental Protection Service EPS 1-WP-77-1. Water Pollution Directorate, Ottawa, ON.

Fjellheim, A. and G.G. Radum. 1990. Acid precipitation: biological monitoring of streams and lakes. Science of the Total Environment. 96: 57-66.

Flannagan, J.F. and D.M. Rosenberg. 1982. Types of artificail substrates used for sampling of freshwater benthic macroinvertebrates. In. Artificial Substrates. Cairns, J. (Ed.). Ann Arbor Science Publ., Ann Arbor Michigan.

Fletcher, W.K. 1981. Analytical methods in geochemical prospecting. In: Govett, G.J.S., Editor, Handbook of exploratation geochemistry, Elsevier, 254pp.

Forstner, U., G.T.W. Wittmann. 1979. Metal pollution in the aquatic environment. Spring-Verlag, New York, N.Y., pp. 486.

Forstner, U. 1982a. Chemical forms of metal enrichment in recent sediments. In. Ore Genesis, Amstutze, G. and others (Eds.), Springer Verlag, New York. pp 191-199.

Forstner, U. 1982b. Accumulative phases for heavy metals in limnic sediments. Hydrobiologia, 91: 269-284.

Forstner, U. 1987. Sediment-associated contaminants - An overview of scientific bases for developing remedial options. In. Ecological Effects of In-Situ Contaminants. Evans, R.T.R., A. Hamilton, M. Munawar, T. Reynoldson, and H. Sadar (eds.), Hydrobiologia 149: 221-246.

Foster, P.L. 1982. Species associations and metal contents of algae from rivers polluted by heavy metals. Freshwater Biology, 12(1): 17-39..

Fortin, D, B. Davis, G. Southam, and T.J. Beveridge. 1995. Biogeochemical phenomena induced by bacteria within sulfidic mine tailings. Journal of Industrial Microbiology 14(2): 178-185.

Fraley, J. J. and B. B. Shephard. 1989. Life history, ecology and population status of bull trout (*Salvelinus confluentus*) in the Flathead Lake and River system, Montana. Northwest Science. 63:133_143.

Frazier, B.E., T.J. Naimo, M.B. Sandheinrich. 1996. Temporal and vertical distribution of total ammonia nitrogen and un-ionized ammonia nitrogen in sediment pore water from the upper Mississippi River. Environmental Toxicology and Chemistry, 15 (2): 92-99.

Freeze, R.A., J.A. Cherry. 1979. Groundwater. Prentice Hall, Englewood Cliffs, N.J.

Frey, D.G. 1966. Limnology in North America. The University of Wisconsin Press, Madison, WI.

Furse, M.T., D. Moss, J.F. Wright, and P.D. Armitage. 1984. The influence of seasonal and taxonomic factors on the ordination and classification of running water sites in Great Britain and on the prediction of their macro-invertebrate communities. Freshwater Biology. 14: 257-280.

Gadsby, J.W., J.A. Malick, S.J. Day. (eds). 1990. Acid Mine Drainage: Designing for closure. BiTech Publishers Ltd.

Galbraith, D.M. 1991. The Mount Washington acid mine drainage reclamation project. In. Second International Conference on the Abatement of Acidic Drainage, Tome 2, Montreal, MEND (Ed.) pp. 145-162.

Gauch, H.G. Jr. 1982. Multivariate analyses in community ecology. Cambridge University Press, London, U.K.

Gelwick, F.P., W.J. Matthews. 1993. Artificial streams for studies of fish ecology. Journal of the North American Benthological Society, 12 (4): 343-347.

Gibb, J.P., R.M. Schuller, R.A. Griffin. 1981. Procedures for the collection of representative water quality data from monitoring wells. Cooperative Groundwater Report 7, Illinois State Water Survey and Illinois State Geological survey, Champaign, IL.

Geiling, W. D., J. R. M. Kelso and E. Iwachewski. 1996. Benefits from incremental additions to walleye spawning habitat in the Current River, with reference to habitat modification as a walleye management tool in Ontario. Can. J. Fish. Aquat. Sci. 53(Suppl. 1):79_87.

Gilbert, R.O. 1987. Statistical methods for environmental pollution monitoring. Van Nostrand Reinhold, New York, NY.

Gilbert, R.O. and R.R. Kinnison. 1981. Statistical methods for estimating the mean and variance from radionuclide data sets containing negative, unreported or less-than values. Health Physics 40: 377-390.

Goede, W.G. and B.A. Barton 1990. organismic indices and an autopsy based assessment as indicators of health and condition of fish. American Fisheries Society Symposium. 8: 93-108.

Golder. 1994. Defining the mixing of effluent from Horseshoe Creek in Hidden Bay. Prepared by Golder Assoc. Prepared for Cameco Corporation, Saskatoon, SK. .

Goldstein, R. 1989. Power and sample size via MS-PC-DOS computers. The American Statistician, 43 (4): 253-260.

Gonzalez, A.M. 1996. A laboratory-formulated sediment incorporating synthetic acid-volatile sulfide. Environmental Toxicology and Chemistry, 15 (12): 2209-2220.

Gordon, N.D., T.A. McMahon and B.L. Finlayson. 1992. Stream hydrology. John Wiley & Sons, Toronto.

Gray, N.F. 1996a. A substrate classification index for the visual assessment of acid mine drainage in lotic systems. Water Research 30(6): 1551-1554.

Gray, N.F. 1996b. Field assessment of acid mine drainage contamination in surface and ground water. Environmental Geology 27(4): 358-361.

Green, R.H. 1979. Sampling design and statistical methods for environmental biologists. John Wiley & Sons.

Green, R.H. 1984. Statistical and nonstatistical considerations for environmental monitoring studies. Environ. Monitor. Assess., 4: 293-301.

Green, R.H. 1989. Inference from observational data in environmental impacts studies: What is valid? What is possible? I.S.I. 47th Session, Paris, August 29-September 6.

Green, R.H. 1993. Application of repeated measures designs in environmental impact and monitoring studies. Australian Journal of Ecology, 18: 81-98.

Green, R.H., J.M. Boyd, and J.S. MacDonald. Relating sets of variables in environmental studies: the sediment quality triad as a paradigm. Environmetrics. 4(4): 439-457. .

Greubel, K., J. Davis, and J. Leckie. 1988. The feasability of using sequential extraction techniques for arsenic and selenium in soils and sediments. Soil Science Society of America Journal. 52: 390-397.

Grubbs, F.E. 1969. Procedures for detecting outlying observations in samples. Technometrics, 11: 1-22.

Guckert, J.B. 1993. Artificial streams in ecotoxicology. Journal of the North American Benthological Society, 12 (4): 350-356.

Gunn, J. M. 1995. Spawing behavior of lake trout: effects on colonization ability. J. Great Lakes Res. 21(Suppl. 1):323_329.

Gunn, J. M., M. Conlon, R. J. Kirk, and S. C. McAughey. 1996. Can trained observers accurately identify lake trout spawning habitat. Can. J. Fish. Aquat. Sci. 53(Suppl. 1):327_331.

Haas, C.N., K. Cidambi, S. Kersten, K. Wright. 1996. Quantitative description of mixture toxicity: Effect of level of response of interactions. Environmental Toxicology and Chemistry, 15 (8): 1429-1437.

Haaparanta, A., E.T. Valtonen, R. Hoffmann, and T. 1996. Do macrophage centres in freshwater fishes reflect the differences in water quality. Aquatic Toxicology (Amsterdam) 34(3): 253-272.

Håkanson, L. 1983. Sediments in aquatic pollution control programmes. In. Principles in Lake Sedimentology, Hakanson, L. and M. Janssen (Eds.) pp 258-282.

Håkanson, L. 1984. Sediment sampling in different aquatic environments: Statistical aspects. Water Resources Research, 20 (1): 41-46.

Håkanson, L. 1992. Sediment Variability. In. Sediment Toxicity Assessment. Burton, G.A. (Ed.). Lewis Publ. Ann Arbor Michigan, pp. 19-34.

Halbert, B.E., J.M. Scharer, R.A. Knapp, D.M. Gorber. 1983. Determination of acid generation rates in pyritic mine tailings. 56th Annual Conf. Of the WPCF, Atlanta, Georgia, U.S.A.

Hall, R.J. 1990. Relative importance of seasonal, short-term pH disturbances during discharge variation on a stream ecosystem. Can. J. Fish. Aquat. Sci. 47: 2261-2274.

Hale, S. S., T. E. McMahon, and P. C. Nelson. 1985. Habitat suitability index models and instream flow suitability curves: chum salmon. Biological Report 82(10.108)..

Hamer 1986. Metallothionein. Annual Review of Biochemistry. 55: 913-951.

Hamilton, S.J. and P.M. Merhle. 1986. Metallothionein in fish: review of its importance in assessing stress from metal contaminants. Transactions of the American Fisheries Society. 115: 596-609.

Hamza-Chaffai, A., R.P. Cosson, C. Amiard-Triquet, A. El Abed. 1995. Physico-chemical forms of storage metals (Cd, Cu and Zn) and metallothionein-like proteins in gills and liver of marine fish from the Tunisian coast: Ecotoxicological consequences. Comp. Biochem. Physiol., 111C (2): 329-341.

Hanna, T.J. 1973. Foundation instrumentation. Trans Tech Publications, Clausthal, Germany.

Hansen, D.J., W.J. Berry, J.D. Mahony, W.S. Boothman, D.M. Di Toro, D.L. Robson, G.T. Ankley, D. Ma, Q. Yan, C.E. Pesch. 1996a. Predicting the toxicity of metal-contaminated field sediments using interstitial concentration of metals and acid-volatile sulfide normalizations. Environmental Toxicology and Chemistry, 15 (12): 2080-2094.

Hansen, D.J., J.D. Mahony, W.J. Berry, S.J. Benyi, J.M. Corbin, S.D. Pratt, D.M. Di Toro, M.B. Able. 1996b. Chronic effect of cadmium in sediments on colonization by benthic marine organisms: An evaluation of the role of interstitial cadmium and acid-volatile sulfide in biological availability. Environmental Toxicology and Chemistry, 15 (12): 2126-2137.

Hare, L. and P.G.C. Campbell 1992. Temporal variations of trace metals in aquatic insects. Freshwater Biology 27(1): 13-27.

Hare, L., R. Carignan, M.A. Huerta-Diaz. 1994. A field study of metal toxicity and accumulation by benthic invertebrates: implications for the acid-volatile sulphide model.

Hare, L., A., Tessier and N. Belzile. 1989. Gut sediments in a burrowing mayfly (*Ephemeroptera, Hexagenia limbata*): their contribution to animal trace element burdens, their removal, and the efficacy of a correction for their presence. Canadian Journal of Fisheries and Aquatic Sciences, 46: 451-456.

Hare, L., A., Tessier, and P.G.C. Campbell. 1991. Trace element distributions in aquatic insects: variations among genera, elements, and lakes. Can. J. Fish. Aquat. Sci. 48(8): 1481-1491.

Harrelson, C.C., C.L. Rawlins and J.P. Potyondy. 1994. Stream channel reference sites. Gen. Tech. Rep. RM-245. Fort Collins, CO: U.S. Dept. Of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 61p.

Harries, J.R., and A.I.M. Ritchie. 1981. The use of temperature profiles to estimate the pyritic oxidation rate in a waste rock dump from an opencut mine. Water, Air and Soil Pollution 15: 405-423.

Harries, J.R., and A.I.M. Ritchie. 1983. Runoff fraction and pollution levels in runoff from a waste rock dump undergoing pyritic oxidation. Water, Air, and Soil Pollution, 19: 155-170.

Harries, J.R. and A.I.M. Ritchie. 1985. Pore gas composition in waste rock dumps undergoing pyritic oxidation. Soil Science 140: 143-152.

Harshbarger, J.C., Spero, P.M., and N.M. Wolcott. 1993. Neoplasms in wild fish from the marine ecosystem emphasizing environmental interactions. *In*. Advances Fishery Science pathobiology of Marine and Estuarine organisms. J.A. Couch and J.W. Fournie (Eds.). pp. 157-177.

Harsham, K.D. 1995. Water sampling for pollution regulation. Gordon and Breach Publishers , Australia.

Hart, B.T. 1982. Uptake of trace metals by sediments ans suspended particulates: a review. Hydrobiologia, 91: 299-313.

Hart, B.T., M.J. Jone, P. Breen. 1983. *In situ* experiments to determine the uptake of copper by aquatic macrophyte *Naja tenuiflora*. Environmental Technology Letters 4: 217-222.

Hart, W.M., K.I. Batarseh, G.P. Swaney, and A.H. Stiller. 1991. A rigoorous model to predict the AMD production rate of mine waste rock. In. Second International Conference on the Abatement of Acidic Drainage, Tome 2, Montreal, MEND (Ed.) pp. 257-270.

Hartman, P.A. 1991. Recent developments and new technologies for microbiological analyses of water - An overview. J.R. Hall, G.D. Glysson, (eds)., American Society for Testing and Materials, Philadelphia.

Harvey , H.H. 1982. population responses of fish in acidified waters. In. Acid Rain / Fisheries. Proceedings of an international symposium on acidic rain and fishery impacts on northeastern North America. . R.E. Johnson (Ed.). Northeastern Division American Fisheries Society.227-242.

Harvey, H.H. and D.A. Jackson. 1995. Acid stress and extinction of a spring spawning fish population. Water, Air and Soil Pollution, 85(2): 383-388.

Hassan, S.M., A.W. Garrison, H.E. Allen, D.M. Di Toro, G.T. Ankley. 1996. Estimation of partition coefficients for five trace metals in sandy sediments and application to sediment quality criteria. Environmental Toxicology and Chemistry, 15 (12): 2198-2208.

Hauer, R.F. 1993. Artificial streams for the study of macroinvertebrate growth and bioenergetics. Journal of the North American Benthological Society, 12 (4): 333-337.

Hawkins, C.P., J.L. Kershner, P.A. Bisson, M.D. Bryant, L.M. Decker, S.V. Gregory, D.A, McCullough, C.K. Overton, G.H. Reeves, R.J. Steedman, and M.K. Young. 1993. A hierarchical approach to classifying stream habitat features. Fisheries 18(6): 3-6.

Hawkins, J.W. and W.W. Aljoe. 1991. Hydrological characteristics of a surface mine spoil aquifer. In. Second International Conference on the Abatement of Acidic Drainage, Tome 1, Montreal, MEND (Ed.) pp. 47-68.

Hayes, M. L. 1983. Active fish capture methods. p. 123-146. In Fisheries Techniques. L. A. Neilsen and D. L. Johnson (Eds.) American Fisheries Society. Bethesda, Maryland.

Heisey, R.M. and A.W.H. Damman. 1982. Copper and lead uptake by aquatic macrophytes in eastern Conneticutt, USA. Aquatic Botany 14: 213-229.

Hellawell, J.M. 1986. Biological indicators of freshwater pollution and environmental management. Elsevier, London.

Helz, G.R. J.H. Dai, R.J. Kipak, N.J. Frendinger, and J.C. Radway. 1987. Processes controlling the composition of acid sulphate solutions evolving from coal. Applied Geochemistry 2: 427-436.

Herasymiuk, G.M., G.W. Wilson, and S.M. Barbour. 1996. Hydrogeologic characterization of a waste rock pile. Presented at the MEND Workshop on Monitoring and Waste Management for Acid Mine Drainage. July 6, 1997, Saskatoon, Saskatchewan.

Herbert, R.B. 1994. Metal transport in groundwater contaminated by acid mine drainage. Nordic Hydrology, 25(3): 193-212.

Herr, C. and N.F. Gray. 1996. Seasonal variation of metal contamination of riverine sediments below a copper and sulphur mine in South-East Ireland. Water Science and Technology, 33(6): 255-261.

Herricks, E.E. and J. Cairns, Jr. 1982. Biological monitoring. Part III - Receiving system methodology baased on community structure. Water Research 16: 141-153.

Hesslein, R.H. 1976. An in-situ sampler for close interval pore water studies. Limnol. Oceanogr. 21: 912-914.

Hesslein, R.H. and E. Slavicek. 1984. Geochemical pathways and biological uptake of radium in small Canadian shiled lakes. Can. J. Fish. Aq. Sci. 41; 459-468.

Hickman, T. and R. F. Raleigh. 1982. Habitat suitability index models: cutthroat trout. U. S. D. I. Fish and Wildlife Service. FWS/OBS_82/10.5.

Higgs, T.W., F.M. Murphy, and C.J. Stewart. 1997. ARD Assessment program from exploration to operation: case studies from the Eskay Creek mine. Proceedings of the 4th International Conference on Acid Rock Drainage. Vol. 1: 385-398.

Hill, T. D. and D. W. Willis. 1994. Influence of water conductivity on pulsed AC and pulsed DC electrofishing catch rates for largemouth bass. N. Amer. J. Fish. Manage. 14: 202-207.

Hildrew, A.G., C.R. Townsend, J. Francis, and K. Finch. 1984. Cellulolytic decomposition in streams of contrasting pH and its relationship with invertebrate community structure. Freshwater Bbiology 14: 323-328.

Hinton, D.E. 1993. Toxicologic histopathology of fishes: a systematic approach and overview. In. Advances in Fishery Science Pathobiology of Marine and Estuary Organisms. J.A. Couch and J.W. Fournie (Eds.). 177-215.

Hinton, D.E. and D.J. Laurén. 1990. Integrative histopathological approaches to detecting effects of environmental stressors on fishes. American Fisheries Society Symposium 8: 51-66.

Hirsch, R.M. 1988. Statistical methods and sampling design for estimating step trends in surface-water quality. Water Resources Bulletin, 24(3): pp.493-503.

Hirner, A., K. Kritsotakis, and H. Tobschall. 1990. Metal-organic associations in sediments - I, comparisons of unpolluted recent and ancient sediments and sediments affected by anthropogenic pollution.. Applied Geochemistry, 5: 491-506.

Hutchison, I.P.G. and R.D. Ellison. 1992. Mine waste management. Prepared for the Canadian Mining Assoc. Lewis Publ. Inc. Chelsea, Michigan, pp 637.

Hodson, P.V., K.R. Munkittrick, R. Stevens, A. Colodey. 1996. A tier-testing strategy for managing programs of environmental effects monitoring. Water Qual. Res. J. Canada, 31 (2): 215-224.

Hoffmann, J.P. 1993. Investigating hydrodynamic effects on diatoms at small temporal and spatial scales. Journal of the North American Benthological Society, 12 (4): 356-359.

Hoke, R.A., J.P. Giesy, J.R. Adams. 1990. Use of linear orthogonal contrasts in analysis of environmental data. Environmental Toxicology and Chemistry, 9: 815-819.

Hopke, P.K. 1992. Factor and correlation analysis of multivariate environmental data. In Methods of Environmental Data Analysis, (ed)., C.N. Hewitt. Elsevier Science Publishers Ltd. pp. 139-180.

Hornbrook, E.H. W. and R.G. Garrett. 1976. Regional geochemical lake sediment survey, east-central Saskatchewan. Geological Survey of Canada, Paper 75-41.

Hornsby, J.K., D Graham, K. Link, and D. Mullins. 1991. Remote sensing to monitor uranium tailings sites - a review. In. Second International Conference on the Abatement of Acidic Drainage, Tome 4, Montreal, MEND (Ed.) pp. 255-274.

Horowitz, A.J. 1991a. A primer on sediment trace-element chemistry, 2nd ed. Lewis Publ. Inc. Chelsea, Michigan.

Horowitz, A.J. 1991b. The role of sediment-trace element chemistry in water-quality monitoring and the need for standard analytical methods. In. Monitoring Water in the 1990's: Meeting New Challenges , Hall, J and D. Glysson (Eds.), ASTM STP 1102, American Society for Testing and Materials, Philadelphia

Horowitz, A.J. and K.A. Elrick. 1986. An evaluation of air elutriation for sediment particle size separation and subsequent chemical analysis. Environmental technology Letters, 7: 17-26.

Horowitz, A.J. and K.A. Elrick. 1988. Interpretation of bed sediment trace metal daat: methods for dealing with the grain size effect. In. Chemical and Biological Characteristics of Sludges, Sediments, Dredge Spoils and Drilling Muds., ASTM STP, Lichtenberg, J.J., J.A. Winter, C.I. Weber, and L. Fradkin (Eds.), American Society for Testing and Materials, Philadelphia. pp. 114-128

Horowitz, A., K. Ellrick, and R. Hooper. 1989. The prediction of aquatic sediment associated trace element concentrations using selected geochemical factors: Hydrological Processess, 3: 347-364.

Howes, B.L., J.W.H. Daecey and S.G. Wakeham. 1985. Effects of sampling technique on measurements of pore water constituents in salt marsh sediments. Limnol. Oceanogr. 30: 221-227.

Huang, P.M., W.K. Liaw. 1977. The nature of a shield lake sediment. Int. Revue ges. Hydrobiol., 62 (3): 329-336.

Huang, P.M., W.K. Liaw. 1978. Distribution and fractionation of arsenic in selected fresh water lake sediments. Int. Revue ges. Hydrobiol., 63 (4): 533-543.

Huang, P.M., W.K. Liaw. 1979. Adsorption of arsenite by lake sediments. Int. Revue ges. Hydrobiol., 64 (2): 263-271.

Hubert, W. A. 1983. Passive capture techniques. p. 95-122. In. Fisheries Techniques. L. A. Neilsen and D. L. Johnson (Eds.). American Fisheries Society. Bethesda, Maryland. pp. 95-122.

Hubert, W.A., R.S. Helzner, L.A. Lee, and P.C. Nelson. 1985. Habitat suitability index models and instream flow suitability curves: arctic grayling riverine populations. U.S. Fish Wild. Serv. Biol. Rep. 82(10.110). 34 p.

Hudson, L.A., J.J.H. Ciborowski. 1996. Teratogenic and genotoxic responses of larval Chironomus salinarius group (Diptera: Chironomidae) to contaminated sediment. Environmental Toxicology and Chemistry, 15 (8): 1375-1381.

Hurlbert,S.H. 1971. The non-concept of species diversity: a critique and alternative parameters. Ecology 52: 577-586.

Hvorslev, J. 1951. Time log and soil permeability in groundwater observations. Waterways Experimental Station, Corps of Engineers, U.S. Army, Bulletin No. 36, Vicksburg, Mississippi, U.S.A.

IHD-WHO (International Hydrological Decade - World Health organization). 1978. Water quality surveys: A guide for the collection and interpretation of water quality data. United Nations Educational, Scientific, and Cultural Organization, Geneva, Switzerland.

Ingersoll, C.G., G.T., Ankley, D.A. Benoit, E.L. Brunson, G.A. Burton, F.J. Dwyer, R.A. Hoke, P.F. Landrum, T.J. Norgerg-King, P.V. winger. 1995. Toxicity and bioaccumulation of sediment-associated contaminants using freshwater invertebrates: A review of methods and applications. Environ. Toxicol. Chem. 14: 1885-1894.

Inskip, P.D. 1982. Habitat suitability index model: northern pike. Western Energy and Land Use Team, Office of Biological Services, Fish and Wildlife Service, U.S. Department of the Interior, Washington, D.C. Biological Report FWS/OBS_82/10.17.

Jackson D.A. and H.H. Harvey. 1995. Gradual reduction and extinction of fish populations in acid lakes. Water, air and Soil Pollution. 85(2): 389-394.

Jearld, A. 1989. Age determination. *In*. Fisheries Techniques. L. A. Neilsen and D. L. Johnson, and S.S. Lampton (Eds.). American Fisheries Society. Bethesda, Maryland pp. 301-324.

Jenne, E., V. Kennedy, J. Burchard, and J. Ball. 1980. Sediment collection and processing for selective extraction and for total metal analysis. In. Contaminants and Sediments Vol 2, Baker, R. (Ed), Ann Arbor Science Publ., Ann Arbor Michigan pp 169-189.

Jiminez, B.D. and J.J. Stegeman. 1990. Detoxification enzymes as indicators of environmental stress on fish. American Fisheries Society Symposium 8: 67-79.

Joern, A., K.D. Hoagland. 1996. In defence of whole-community bioassays for risk assessment. Environmental Toxicology and Chemistry, 15 (4): 407-409.

Johnson, D.L. 1983. Sampling considerations p. In. Fisheries Techniques. L. A. Neilsen and D. L. Johnson (Eds.) . American Fisheries Society. Bethesda, Maryland. pp. 1-22.

Jonasson, I. 1977. Geochemistry of sediment/water interactions of metals, including observations on availability. In. The Fluvial Transport of Sediment-Associated Nutrients and Contaminants. Shear, H. and A. Watson (Eds.). IJC/PLUARG, Windsor, Ontario, pp 255-271.

Johnson, M. G., J. H. Leach, C. K. Minns, and C. H. Olver. 1977. Limnological characteristics of Ontario lakes in relation to associations of walleye (*Stizostedion vitreum*), northern pike (*Esox lucius*), lake trout (*Salvelinus namaycush*), and smallmouth bass)*Micropterus dolomieui*). J. Fish. Board Res. Can. 34:1592-1601.

Johnson, R.K. and T. Wiederholm. 1989. Classification and ordination of profundal macroinvertebrate communities in nutrient poor, oligo meso-humic lakes in relationship to environmental data. Freshwater Biol. 21: 375-386.

Jones, G.J. 1982. Activities of aerobic and anaerobic bacteria in lake sediments and their effect on the water column. (eds)., D.B. Nedwell, C.M. Brown, Academic Press.

Jones, K.C. 1986. The distribution and partitioning of silver and other heavy metals in sediments associated with an acid mine drainage stream. Environmental Pollution (Series B), 12: 249-263.

Jones B. and C. Bowser. 1978. The mineralogy and related chemistry of lake sediments. In. Lakes: Chemistry, Geology, Physics. A. Lerman (Ed.), Springer Verlag, New York. pp 179-235.

Käji, J.H.R. anbd Y. Kojima. 1987. Chemistry and biochemistry of metallothionein. Experientia Supplementum Metallothionein II 52: 25-61.nisrty and Kojima 1987

Karr, J.R. 1987. Biological Monitoring and Environmental Assessment: a Conceptual Framework. Environmental Management, 11(2): 249-256.

Katapodis, C. 1992. Introduction to fishway design. Freshwater Institute, Department of Fisheries and Oceans.

Keith, L.H. 1991. Environmental sampling and analysis: A practical guide. Lewis Publishers, MI.

Kelly, M., W.J. Allison, A.R. Garman, C.J. Symon. 1988. Mining and the freshwater environment. Elsevier Science Publishers Ltd.

Kerekes, J., B. Freedman, G. Howell, and P. Clifford. 1984. Comparison of the characteristics of an acidic eutrophic and an acidic oligotrophic lake near Halifax, Nova Scotia. Water Pollut. Res. J. Can., vol. 19: 1-10.

Kheboian, C and C. Bauer. 1987. Accuracy of seltive extraction procedures for metal speciation in model aquatic sediments. Analytical Chemistry, 59: 1417-1423.

Keough, M.J., B.D. Mapstone. 1997. Designing environmental monitoring for pulp mills in Australia. Water Science and Technology, .35(2): 397-3398.

Kiffney, P.M., W.H. Clements. 1996. Size-dependent response of macroinvertebrates to metals in experimental streams. Environmental Toxicology and Chemistry, 15 (8): 1352-1356.

King, A., and A.N. Sartorelli. 1991. Mapping acidified groundwater using surface geophysical methods. In. Second International Conference on the Abatement of Acidic Drainage, Tome 3, Montreal, MEND (Ed.) pp. 551-487.

Kinnlburgh, D.G. and D.L. Miles. 1983. Extraction and chemical analysis of interstitial water from soils and rocks. Environ. Sci. Technol., 17: 362-368..

Kitchell, J. F., M. G. Johnson, C. K. Minns, K. H. Loftus, L. Grieg, and C. H. Olver. 1977. Percid habitat: the river analogy. J. Fish. Res. Board Can. 34: 1936-1940.

Klaverkamp, J.F. and D.A. Duncan. 1987. Acclimation to cadmium toxicity by white suckers: cadmium binding capacity and metal distribution in gill and liver cytosol. Environmental Toxicology and Chemistry, 6: 275-289.

Klaverkamp, J.F., W.A. Macdonald, D.A. Duncan, and R. Wageman. 1984. Metallothionein and acclimation to heavy metals in fish: a review. In. Contaminants Effects on Fisheries. Cairns, V.W., P.V. Hodson, and J.O. Nriagu. (EDS.).W iley, New York.

Klaverkamp, J.F., M.D. Dutton, H.S. Majewski, R.V. Hunt, and L.J. Wesson. 1991. Evaluating the effectiveness of metal pollution controls in a smelter by using metallothionein and other biochemical responses in fish. In. Metal Ecotoxicology: Concepts and Applications. M.C. Newman and A.W. McIntosh (Ed.s). Lewis Publ. Chelsea, Michigan. pp. 33-64.

Klemm, D.J., P.A. Lewis, F. Fulk, J.M. Laorchak. 1990. Macroinvertebrate field and laboratory methods for evaluating the biological integrity of surface waters. EPA/600-4-90/030. Environmental Monitoring Systems Laboratory, Office of Research and Development, Cincinnati, OH.

Klute, A. (Ed). 1986. Methods of soil analysis. Part I. Physical and Mineralogical Methods. Second Edition. Number 9 (Part 1) in the series Agronomy. American Society of Agronomy, Soil, Science Society of America, Madison, WI.

Kondos, P.D., R.J.C. MacDonald, and J.M. Zinck. 1991. Studies on the removal of heavy metals from acidic mineral effluents. In. Second International Conference on the Abatement of Acidic Drainage, Tome 2, Montreal, MEND (Ed.) pp. 47-70.

Krajca, J.M. 1989. Water sampling. Ellis Horwood Ltd. West Sussex, England.

Krantzberg, G. 1989. Accumulation of essential and nonessential metals by chironomid larvae in relation to physical and chemical properties of the elements. Can. J. Fish Aquat. Sci., 46: 1755-1761.

Krieger, D. A., J. W. Terrell, and P. C. Nelson. 1983. Habitat suitability index models: yellow perch. FWS/OBS 82/10.55..

Kromm, M.D., P. Davison, H. Zhang, W. Davison. 1994. High resolution pore-water sampling with a gel sampler. Limnol. Oceanogr. 39(8): 1967-1972.

Kwong, Y.T.J. 1991. Acid generation in waste rock as exemplified by the Mount Washington minesite, British Columbia, Canada. In. Second International Conference on the Abatement of Acidic Drainage, Tome 1, Montreal, MEND (Ed.) pp. 175-190.

Kwong, Y.T.J. and W.G. Whitley . 1992. Natural acid rock drainage at MacMillan Pass, Yukon. National Hydrological Research Inst. Saskatoon, SK., Contribution No. 92046.

Kwong, Y.T. and K.D. Ferguson. 1997. Mineralogical changes during NP determination and their implications. Proceedings of the 4th International Conference on Acid Rock Drainage. Vol. 1: 435-448.

Lafleur, R.E., D. Roy, D., Couillard, and R. Guay. 1993. Determination of iron oxidizing bacteria numbers by a modified MPN procedure. In. Biohydrometallurgical Technologies. A.E. Torma, J.E. Wey, and V.L. Lakshmanan (Eds). The Minerals, Metals and Materials Society, 433-442.

Lam, D.C.L., G. Bowen, J.P. Kerby, D.A. Swayne. 1991. A water quality advisor for ranking mine effluent and toxicity data. *In*. Second International Conference on the Abatement of Acidic Drainage, Tome 2, Montreal, MEND (Ed.) pp. 391-408.

Lamberti, G.A. 1993. Grazing experiments in artificial streams. Journal of the North American Benthological Society, 12 (4): 337-342.

Lamberti, G.A. and A.D. Steinman. 1993. Introduction. In. Research in Artificial Streams: applications, uses and abuses. Journal ofr the North American Benthological Society, 12: 314-317.

Lambeth, R.H., B.M. Stewart, and B.C. Williams. 1991. An investigation of hydrogeochemical mechanisms in an abandoned sulphide tailings impoundment and underlying aquifer. In. Second International Conference on the Abatement of Acidic Drainage, Tome 3, Montreal, MEND (Ed.) pp. 43-60.

Lasorsa, B. and Casa, A. 1996. A comparisonn of sample handling and analytical methods for determination of acid volatile sulfides in sediment. Marine Chemistry 52: 211-220.

Lawrence, R.W. and Y. Wang. 1997. Determination of neutralization potential in the prediction of acid rock drainage. Proceedings of the 4th International Conference on Acid Rock Drainage. Vol. 1: 449-464.

Legendre, P. and M. Fortin. 1989. Spatial pattern and ecological analysis. Vegetatio 80: 107-138.

Lee, G., M.R. Ellersieck, F.L. Mayer, and G.F. Krause. 1995. Predicting chronic lethality of chemicals to fishes from acute toxicity test data; multifactor probit analysis. Environmetnal Toxicology and Chemistry, 14(2): 345-349.

Lefebvre, R., P. Gélinas, and D. Isabel 1994. Heat transfer during acid mine production in a waste rock dump, la Mine Doyon (Quebec). Prepared for MEND, Report 1.14.2, March.

Leland, H.V., S.V. Fend, T.L. Dudly, and J.L. Carter. 1989. Effects of copper on species composition of benthic insects in a Sierra Nevada, California, stream. Freshwater Biol. 21: 163-179.

Lemly, D.A. 1993. Metabolic stress during winter increases the toxicity of selenium to fish. Aquatic Toxicology, 27: 133-158.

Leonard, E.N., G.T. Ankley R.A. Hoke. 1996. Evaluation of metals in marine and freshwater surficial sediments from the environmental monitoring and assessment program relative to proposed sediment quality criteria for metals. Environmental Toxicology and Chemistry, 15 (12): 2221-2232.

Lester, N.P., W.I. Dunlop,and C.C. Willox. 1996. Detecting changes in the near shore fish community. Canadian Journal of Fisheries and Aquatic Science, 53(Suppl.1): 391-402.

Levinson, A.A. 1980. Introduction to exploration geochemistry, second edition. Applied Publishing Ltd. Wilmette, Illinois, U.S.A.

Liabastre, A.A., K.A. Carlberg, M.S. Miller. 1992. Quality assurance for environmental assessment activities. In: Methods of Environmental Data Analysis, (ed)., C.N. Hewitt. Elsevier Science Publishers Ltd. pp. 259-299.

Liaw, W.K. 1991. Habitat suitability criteria for walleye spawning and egg incubation in Saskatchewan. Saskatchewan Fisheries Laboratory Dept. of Parks and Renewable Resources.

Liber, K., D.J. Call, T.P. Markee, K.L. Schmude, M.D. Balcer, F.W. Whiteman, G.T. Ankley. 1996. Effects of acid-volatile sulfide on zinc bioavailability and toxicity to benthic macroinvertebrates: A spiked-sediment field experiment. Environmental Toxicology and Chemistry, 15 (12): 2113-2125.

Liebetrau, A.M. 1979. Water quality sampling: Some statistical considerations. Water Resources Research, 15: 1717-1725.

Lindahl, L.A. 1991. Who's going to pay the bill? - regulatory aspects and financing of remedial action in Sweden. In. Second International Conference on the Abatement of Acidic Drainage, Tome 2, Montreal, MEND (Ed.) pp. 119-126.

Lindgren, B.W., G.W. McElrath, D.A. Berry. 1978. Introduction to probability and statistics. Collier Macmillan Ltd. New York, NY.

Lobel, P.B., S.P. Belkhode, S.E. Jjackson and H.P. Longerich. 1991. Sediment in the intestinal tract: a potentially serious source of error in aquatic biological monitoring programs. Mar. Environ. Res. 31: 163-174.

Long, B.R. and P.M. Chapman. 1985. A sediment quality triad: measures of sediment contamination, toxicity and infaunal composition in Puget Sound. marine Pollution Bulletin 16: 405-415.

Loring, D. 1990. Lithium - a new approach for the granulometric normalization of trace metal data. Marine Chemistry, 29: 155-168.

Loring, D. and R.T.T. Rantala. 1992. Manual for the geochemical analysis of of marine sediments and suspended particulate matter. Earth Science Review, 32: 235-283.

Lovich, J.E. and R.E. Lovich. 1996. The decline of native brook trout (*Salvelinus fontinalis*) populations along the upper West Branch of the Susquehanna River: canaries outside the coal mine. Journal of the Pennsylvania Academy of Science70(2): 55-60.

Lowell, R.B., J.M. Culp, F.J. Wrona. 1995. Stimulation of increased short-term growth and development of mayflies by pulp mill effluent. Environmental Toxicology and Chemistry, 14 (9): 1529-1541.

Lowell, R.B., J.M. Culp, F.J. Wrona. 1995. Toxicity testing with artificial streams: Effects of differences in current velocity. Environmental Toxicology and Chemistry, 14 (7): 1209-1217.

Lowson, R.T. B.J. Reedy and J.K. Beattie 1991. An $^{18}O_2$ tracer study of pyritic oxidation. In. Second International Conference on the Abatement of Acidic Drainage, Tome 4, Montreal, MEND (Ed.) pp. 159-172..

Lundgren, D.G. and M. Silver. 1980. Ore leaching by bacteria. Annual Review of Microbiology. 34: 263-283.

Luoma, S.N. 1983. Bioavailability of trace metals to aquatic organisms - A review. The Science of the Total Environment, 28: 1-22.

Luoma, S.N. 1989. Can we determine the biological availability of sediment-bound trace elements? Hydrobiologia, 176/177: 379-396.

Lyne, T.B., J.W. Bickham, T. Lamb, J.W. Gibbons. 1992. The application of bioassays in risk assessment of environmental pollution. Risk Analysis, 12 (3): 361-365.

Lyon, J. 1992. The length of stream to sample with a towed electrofishing unit when fish species richness is estimated. N. Amer. J. Fish. Manage. 12:198-203.

MacFarlane, R.D., G.L. Bullock, J.J.A. McLaughlin. 1986. Effects of five metals on susceptibility of striped bass to flexibacter columnaris. Transactions of the American Fisheries Society, 115: 227-231.

Mackay, R.J. and K.E. Kersey. 1985. A preliminary study of aquatic insect communities and leaf decomposition in acid streams near Dorset, Ontario. Hydrobiologia 122: 3-11.

Mackey, A.P. and S. Mackey. 1996. Spatial distribution of acid-volatile sulphide concentration and metal bioavailability in mangrove sediments from the Brisbane River, Australia. Environmental Pollution 93(2): 205-209.

Mackay, W.C., G.R. Ash, H.J. Norris. 1990. Fish ageing methods for Alberta. R.L. & L. Environmental Services Ltd., Edmonton, Alberta.

MacKnight, S.D. 1994. Selection of bottom sediment sampling stations. In. Handbook of Techniques for Aquatic Sediment Sampling 2nd Ed. Mudroch, A. and S.D. MacKnight (Eds.). Lewis Publ. , Chelsea, Michigan.

Maccubbin, A.E. and N. Ersing. 1991. Tumors in fish from the Detroit River. Hydrobiologia 219: 301-306.

MacFarlane, R.D., G.L. Bullock, and J.J.AMcLaughlin. 1986. Effects of five metals on susceptibility of striped bass to *Flexibacter columnaris*. Transactions of the American Fisheries Society. 115: 227-231.

Mahony, J.D., D.M. Di Toro, A.M. Gonzalez, M. Curto, M. Dilg, L.D. De Rosa, L.A. Sparrow. 1996. Partitioning of metals to sediment organic carbon. Environmental Toxicology and Chemistry, 15 (12): 2187-2197.

Malcolm, S.J., and S.O. Stanley. 1982. The sediment environment. (Eds)., D.B. Newell, C.M. Brown, Academic Press.

Malley, D.F., J.F. Klaverkamp, S.B. Brown and P.S.S. Chang. 1993. Increase in metallothionein in freswater mussles Anodonta grabdis exposed to cadmium in the laboratory and the field. Water pollut. Res. Can. 28(1): 253-273.

Mallatt, J. 1985. Fish gill structural changes induced by toxicants and other irritants: A statistical review. Can. J. Fish Aquat. Sci., 42: 630-648.

Mapstone, B.A. 1995. Scalable decision rules for environmental impact studies: effect size, Type I and Type II errors. Ecological Applications 5: 401-410.

Marcus, M. D., W. A. Hubert, and S. H. Anderson. 1984. Habitat suitability index models: lake trout (exclusive of the Great Lakes). U. S. Fish Wildl. Serv. FWS/OBS_82/10.84. 12 pp.

Marshall, T. R. 1996. A hierarchical approach to assessing habitat suitability and yield potential of lake trout. Can. J. Fish. Aquat. Sci. 53(Suppl. 1):332_341.

Masters, J.A., M.A. Lewis, D.H. Davidson, and R.D. Bruce. 1991. Validation of a four-day Ceriodaphnia toxicity tests and statistical considerations in data analysis. Environ. Toxicol. Chem. 10: 47-55

Martin, N. V., and C. H. Olver. 1980. The lake charr, *Salvelinus namaycush*. Pages 205_277 in E. K. Balon (ed.) Charrs _ salmonid fishes of the genu *Salvelinus*. Junk Publ., The Hague, Netherlands.

Matthews, R.A., W.G. Landis, G.B. Matthews. 1996. The community conditioning hypothesis and its application to environmental toxicology. Environmental Toxicology and Chemistry, 15 (4): 597-603.

Mayes, R.A., A.W. McIntosh, V.L. Anderson. 1977. Uptake of cadmium and lead by a rooted aquatic macrophyte (Elodea canadensis). Ecology 58: 1176-1180.

McCarrraher, D. B. and R. E. Thomas. 1972. Ecological significance of vegetation to northern pike, *Esox lucius*, spawning. Trans. Amer. Fish. Soc. 101:560-563.

McCormick P.V. 1993. Investigating mechanisms of longitudinal interactions. Journal of the North American Benthological Society, 12 (4): 366-370.

McFarlane, G.A. and W.G. Franzin. 1978. Elevated heavy metals: A stress on a population of white suckers, (*Catostomus commersoni*), in Hamell Lake, Saskatchewan. J. Fish. Res. Board Can., 35: 963-970.

McIntire, D.C. 1993. Historical and other perspectives of laboratory stream research. Journal of the North American Benthological Society, 12 (4): 318-323.

McLeod, A.I., K.W. Hipel. 1991. Trend analysis methodology for water quality time series. Report prepared for Environment Ontario, Research and Technology Branch, Rexdale, Ontario by McLeod-Hipel and Assoc., Waterloo, Ontario, pp. 67 + App.

McMahon, T. E. 1982. Habitat suitability index models: creek chub. FWS/OBS 82/10.4.

McMahon, T. E. 1983. Habitat suitability index models: coho salmon. FWS/OBS 82/10.49.

McMahon, T.E., J.W. Terrell and P.C. Nelson. 1984. Habitat suitability information: Walleye. U. S. Fish and Wildlife Service Biological Service Program FWS/OBS - 82/10.56. 43 pp.

MEND 1.11.1. 1991. Critical literature review of acid drainage from waste rock.

MEND 1.14.1. 199? Critical review of geochemical processes and geochemical models adaptable for prediction of acidic drainage from waste rock. Prepared by Noranda Technology Centre in collaboration with Alberta Research Council and University of Western Ontario.

MEND 1.14.2. 1997. Sampling and monitoring data from the Doyon South Waste Rock Dump (1990-1995) (diskette included), March 1994, revised sept. 1997.

MEND 1.14.2a. 1994. Diversité microbiologique dans la production de drainage minier acide à la halde sud de la Mine Doyan, March.

MEND 1.14.2b. 1994. Development of a modified MPN procedure to enumerate iron oxidizing bacteria, March. Prepared by R. Guay.

MEND 1.14.2c. 1994. Heat transfer during acid mine drainage production in a waste rock dump, La Mine Doyon (Quebec).

MEND 1.14.2d. 1994. Monitoring of acid mine drainage: Chemical data from La Mine Doyon (Quebec).

MEND 1.14.2e. 1994. Water budget for the waste rock dump at La Mine Doyon (Quebec).

MEND 1.14.2f. 1994. Mineralogical transformations assocaited with AMD production in a waste rock dump - La Mine Doyon (Quebec).

MEND. 1.14.2g. 1994. Monitoring and Modeling of acid mine drainage from waste rocks dumps: La Mine Doyon case study. Prepared by, P. Gelinas, R. Lefebvre, M. Choquette, D. Isabel, J. Locat, R. Guay.

MEND 1.16.1a. 1989. Investigation of prediction techniques for acid mine drainage.

MEND. 1.16.1b. 1991. Acid rock drainage prediction manual: A manual of chemical evaluation procedures for the prediction of acid generation from mine wastes. Prepared by, Coastech Research INC.

MEND. 1.16.1c. 1991. New methods of determination of key mineral species in acid generation prediction by acid-base accounting. Prepared by Norecol Environmental Consultants Ltd.

MEND 1.16.3. 1996. Determination of neutralization potential for acid rock drainage prediction. , July 1996.

MEND1.17.1a 1986. Hydrogeochemical investigation of reactive tailings at the Waite Amulet Tailings Site, Noranda, Quebec - 1985 Program.

MEND1.17.1b 1987. Hydrogeochemical investigation of reactive tailings at the Waite Amulet Tailings Site, Noranda, Quebec, Phase 2 - 1986 Program.

MEND1.17.1c 1988. Hydrogeochemical investigation of reactive tailings at the Waite Amulet Tailings Site, Noranda, Quebec, Phase 3 - 1987 Program, Volumes I, II

MEND1.17.1c 1989. Hydrogeochemical investigation of reactive tailings at the Waite Amulet Ttailings Site, Noranda, Quebec, Phase 3 - 1987 Program, Supplementary Report. .

MEND1.17.1d 1990. Hydrogeochemical investigation of reactive tailings at the Waite Amulet tailings Site, Noranda, Quebec, Generation and evolution of acidic porewaters at the Waite Amulet Tailings, Final Report. .

MEND. 1.19.1. 1994. Final report: Long term acid generation studies: Cinola project British Columbia. Prepared by, Norecol, Dames & Moore.

MEND. 1.21.1a. 1990. Critical review of the reactive acid tailings assessment program. Prepared by, SENES Consultants Limited and BEAK Consultants Limited.

MEND 1.22.1a. 1994. Field procedures manual: gas transfer measurements waste rock piles Health Steel Mines New Brunswick.

MEND. 1.25.1. 1994. Soilcover users manual version 1.0. Prepared by, Geotechnical Engineering Group Department of Civil Engineering College of Engineering University of Saskatchewan.

MEND. 1.32.1. 1993. Prediction and prevention of acid rock drainage from a geological and mineralogical perspective. Prepared by, Y.T. John Kwong.

MEND. 2.11.1a. 1989. Subaqueous disposal of reactive mines wastes: An overview. Prepared by, Rescan Environmental Services Ltd.

MEND 2.11.1a 1990a. A preliminary assessment of subaqueous tailings disposal in Benson Lake, British Columbia.

MEND 2.11.1a 1990b. A Preliminary Assessment of subaqueous Tailings Disposal in Anderson Lake,Manitoba.

MEND 2.11.1a 1990c. A Preliminary Assessment of subaqueous Tailings Disposal in Mandy Lake, Manitoba.

MEND 2.11.1a 1990d A Preliminary Assessment of subaqueous Tailings Disposal in Buttle lake, British Columbia.

MEND. 2.11.1c. 1991. A preliminary biological and geological assessment of subaqueous tailings disposal in Benson Lake, British Columbia. Prepared by, Rescan Environmental Services Ltd.

MEND. 2.11.1c. 1992. Chemical diagenesis of submerged mine tailings in Benson Lake and Natural sediments in Keogh Lake, Vancouver Island, British Columbia. Prepared by, Rescan Environmental Services Ltd.

MEND. 2.11.1d. 1992. A critical review of MEND studies conducted to 1991 on subaqueous disposal of tailings. Prepared by, The Rawson Academy of Aquatic Science.

MEND. 2.11.2a. 1993. Literature review report: Possible means of evaluating the biological effects of sub-aqueous disposal of mine tailings, final report. Prepared by, P.C.G. Campbell, R. Roy.

MEND. 2.13.2b. 1994. Flooding of a mine tailing site suspension of solids - impact and prevention. Prepared by, Geotechnical Research Centre.

MEND. 2.20.1. 1994. Final report: Evaluation of alternate dry covers for the inhibition of acid mine drainage from tailings. Prepared by, SENES Consultants Limited.

MEND. 2.21.1. 1991. Development of laboratory methodologies for evaluating the effectiveness of reactive tailings covers. Prepared by, Ernest K.Yanful, Ph.D.

MEND. 2.24.1. 1989. Preparation of a manual of methods used in revegetation of reactive sulphide tailings basins. Prepared by, B.W. Brooks, Dr. T.H. Peters, Dr. J.E. Winch.

MEND. 2.25.1a. 1992. Reclamation of sulphide tailings using municipal solid waste compost: Literature review and recommendations. Prepared by, W. Glenn, Pierce Ph.D.

MEND. 2.31.1a. 1992. Heath Steel waste rock study, Phases 1 - 3.

MEND. 2.31.1b. 1994. Engineering design and construction report phase IV - composite soil cover acid waste rock study Heath Steele mines. Prepared by, ADI Nolan Davis.

MEND. 2.35.2a. 1993. Evaluation of techniques for preventing acidic rock drainage: First milestone report. Prepared by, Ernest K. Yanful and Serge C. Payant.

MEND. 2.36.1. 1995. Final report: Review of in-pit disposal practices for the prevention of acid drainage - case studies volume 1 of 2. Prepared by, SENES Consultants Limited.

MEND. 2.44.1. 1992. Microbial plugging of uranium mine tailings to prevent acid mine drainage - final report. Prepared by, Environmental Laboratory.

MEND. 3.11.l. 1993. Treatment of acidic seepages using wetland ecology and microbiology: Overall program assessment, final report. Prepared by, M. Kalin.

MEND. 3.12.1. 1990. Final report: Assessment of existing natural wetlands affected by low pH, metal contaminated seepages (acid mine drainage). Prepared by, P. Lane and Associates Limited.

MEND. 3.12.2. 1993. Panel wetlands - a case history of partially submerged pyritic uranium tailings under water. Prepared by, N.K. Dave.

MEND. 3.21.1a. 1991. Study on metals recovery/recycling from acid mine drainage. Prepared by, Mineral Sciences Laboratories.

MEND. 3.22.1. 1994. Canada-wide survey of acid mine drainage characteristics. Prepared by, Mineral Sciences Laboratories, L.J. Wilson.

MEND. 3.32.1. 1994. Acid mine drainage - status of chemical treatment and sludge management practices. Prepared by, SENES Consultants Limited.

MEND. 4.1.1. 1989. Field sampling manual for reactive sulphide tailings. Prepared by, CANECT Environmental Control Technologies Limited.

MEND. 4.2.1. 1993. Report on: Review of Canadian and United States legislation relevant to decommissioning acid mine drainage sites. Prepared by, Jacques Whitford Environment Limited.

MEND. 4.5.1-1. 1994. Review of waste rock sampling techniques. Prepared by, SENES Consultants Limited.

MEND. 4.5.1-2. 1994. Handbook for waste rock sampling techniques. Prepared by, SENES Consultants Ltd.

MEND. 4.6.1. 1994. Applications of geophysical methods for monitoring acid mine drainage. Prepared by, A. King, INCO Exploration and Technical Services Inc.

MEND. 4.6.3. 1994. Application of remote sensing and geophysics to the detection and monitoring of acid mine drainage.

MEND. 4.8.2. 1994. Evaluation of an underwater monitoring probe for locating and estimating the impact of groundwater discharges to surface waters adjacent to potential sources of acid mine drainage. Prepared by D.R. Lee and r.D. Bianco, AECL Research, Chalk River Laboratories.

MEND. 5.8.1. 1995. Economic evaluation of acid mine drainage technologies. Prepared by, Geocon.

MEND. 6.1. 1993. Preventing AMD by disposing of reactive tailings in permafrost: MEND new ideas projects. Prepared by, Geocon.

MEND. 6.2. 1994. Polymer-modified clay as impermeable barriers for acid mining tailings. Prepared by, S. Shou, B. Wiwchar, W.D. Gunter, D. Gao.

MEND. 6.3. 1994. A critique of Japanese acid mine drainage technology. Prepared by, Frank E. Smith.

MEND 1997 (In Press). Whistle Mine. Waste rock pile monitoring. Laboratory and field measurements. Prepared by Golder Associates Ltd. and SENES Consultants Ltd. .

Merkowsky, J.J. 1989. Assessment of arctic grayling populations in northern Saskatchewan. Saskatchewan Parks and Renewable Resources, Fisheries Technical Report 89-4.

Mersch, J., P. Wagner, and J.C. Pihan. 1996. Copper in indigenous and transplanted zebra mussels in relation to changing water concentrations and body weight. Environmental Toxicology and Chemistry, 15 (6): 886-893.

Merritt R.W. and K.W. Cummins. 1984. An introduction to the aquatic insects of North America, 2nd Ed. Kendell Hunt Publ., Dubuque, IA.

Milani, D., K.E. Day, D.J. McLeay, and R.S. Kirby. 1996. recent intra- and interlaboratory studies related to the devlopment and standardization of Environment Canada's biological test methods for measuring sediment toxicity using freshwater amphipods (Hyallella azteca) or midge larvae (Chironomus riparius). technical report prepared by National Water research Institute Burlington, Ont.. Prepared for Environment Canada Method Development and Application Section.

Miller, G.C. and D. Eldon. 1991. Financial assurance for mine reclamation, decommissioning and post-closure obligations. In. Second International Conference on the Abatement of Acidic Drainage, Tome 2, Montreal, MEND (Ed.) pp. 127-144.

Miller, P.A., K.R. Munkittrick, and D.G. Dixon. 1992. Relationship between concentrations of copper and zinc in water, sediment, benthic invertebrates, and tissues of white sucker (*Catostomus commersoni*) at metal contaminated sites.

Miller P.A., R.P. Lanno, M.E. McMaster, D.G. Dixon. 1993. Relative contributions of dietary and waterborne copper to tissue copper burdens and waterborne-copper tolerance in rainbow trout (Oncorhynchus mykiss). National Research Council, 50 (8): 1683-1689.

Miller, R.A. 1996. Montana Department of environmental quality guidelines for mine rock and resource characterization during the exploration and feasibility stages of mineral project development, draft. Hard Rock Bureau, Montana Department of Environmental Quality.

Miller, S.D., J.J. Jeffrey, and J.W.C. Wong. 1991a. In-pit identification of acid forming waste rock at the Golden Cross gold mine, New Zealand. In. Second International Conference on the Abatement of Acidic Drainage, Tome 3, Montreal, MEND (Ed.) pp. 137-152.

Miller, S.D., J.J. Jeffrey, and J.W.C. Wong. 1991b. Use and misuse of the acid-base account for AMD prediction. In. Second International Conference on the Abatement of Acidic Drainage, Tome 3, Montreal, MEND (Ed.) pp. 489-506.

Miller, S., A. Robertson, and T. Donohue. 1997. Advances in acid drainage prediction using the net acid generation (NAG) test. Proceedings of the Fourth International Conference on Acid Rock Drainage. 551-566.

Minns, C. K., J. E. Moore and V. W. Cairns. 1990. Current and potential adult habitat of northern pike (*Esox lucius*) in Hamilton Harbour, Lake Ontarion. *In* Proceedings of the National Conference on GIS for the 1990s, 5_8 March 1990, Ottawa, Ont. pp. 1020_1029.

Minns, C. K., R. G. Randall, J. E. Moore and V. W. Cairns. 1996. A model simulating the impact of habitat supply limits on northern pike, *Esox lucius*, in Hamilton Harbour, Lake Ontario. Can. J. Fish. Aquat. Sci. 53(Suppl. 1):20-34.

Mitsch, W.J., J.G. Gosselink. 1993. Wetlands Ed. 2. Van Nostrand Reinhold, New York, NY.

Morin, A. 1985. Variability of density estimates and the optimization of sampling programs for stream benthos. Can. J. Fish. Aquat. Sci., 42: 1530-1534.

MOE (Ministry of Environment and Energy, Ontario) 1993a. Development of the Ontario provincial sediment quality guidelines for arsenic, cadmium, chromium, copper, lead, manganese, mercury, nicle and zinc. Prepared by R. Jaagumagi Water Resouces Branch, Ontario, log 92-2309-069.

MOE. 1993b. Sediment assessment: a guide to study design, sampling and laboratory analysis. Prepared by R. Jaagumagi and D. Persaud, Water Resouces Branch, Ontario, PIBS 2595E.

Moline, G.R., J.J. Beauchamp, and T. Wright. 1996. Determining an optimal sampling frequency for measuring bulk temporal changes in ground-water quality. Ground Water 34(4): 579-587.

Morin, K.A. 1988. Physical and chemical hydrogeology of uranium tailings in Canada and the United States of America. In. Proceeding of the International Groundwater Symposium, International Association of Hydrogeologists, Halifax Nova Scotia. C.L. Lin (Ed.), pp 175-188.

Morin, K.A. and J.A. Cherry. 1988. Migration of acidic groundwater seepage from uranium tailings impoundments, 3: Simulation of the conceptual model with application to seepage area A. Journal of Contaminant Hydrology, 2: 323-342.

Morin, K.A. and N.M. Hutt. 1993. The use of routine monitoring data for assessment and prediction of water chemistry. Proceedings of the 17th Annual Mine Reclamation Symposium. Port Hardy British Columbia. Mining Assocaition of British Columbia. Pp 191-201.

Morin, K.A. and N.M. Hutt. 1995. Prediction of future water chemistry from from Island Copper Mine's on-land-dumps. Proceedings of the 19th Annual Mine Reclamation Symposium. Port Hardy British Columbia. May 4 - 7 Mining Assocaition of British Columbia.

Morin, K.A., J.A. Cherry, N.K. Dave, T.P. Lim, and A.J. Viryrka. 1988. Migration of acidic groundwater seepage from uranium tailings impoundments, 1: Field study and conceptual hydrogeochemical model. Journal of Contaminant Hydrology, 2: 271-203.

Morin, K.A., E. Gerencher, C.E. Jones, D.E. Konasewich, J.R. Harries. 1990. Draft, Critical literature review of acid drainage from waste rock. MEND, pp. 1-160.

Morin, K.A. N.M. Hutt, and R. McArthur. 1995. Statistical assessment of past water chemistry to predict future chemistry at Noranda Minerals' Bell Mine 1995. Paper presented at Sudbury '95. Conference on Mining and Environment, Sudbury, Ontario.

Morrison, R.D. 1983. Ground water monitoring technology, procedures, equipment and applications. TIMCO Mfg. Inc., Prairie Du Sac, WI.

Mudroch, A. 1984. Particle size effects on concentration of metals in Lake Erie bottom sediments. Water Pollution Research Journal of Canada, 19: 27-35.

Mudroch, A. and S.D. MacKnight. 1991. Handbook of techniques for aquatic sediment sampling. CRC Press, Boca Raton, Florida.

Mudroch, A, and J.M. Azcue. 1995. Manual of aquatic sediment sampling. Lewis Publ. London.

Munch, G.J. and R.W.D. Killey. 1985. Equipment and methodology for sampling and testing non-cohesive sediments. Ground Water Monitoring Review 5(1): 38-42.

Munkittrick, K.R., D.G. Dixon. 1988a. Evidence for a maternal yolk factor associated with increased tolerance and resistance of feral white sucker (Catostomus commersoni) to waterborne copper. Ecotoxicology and Environmental Safety, 15: 7-20.

Munkittrick, K.R., D.G. Dixon. 1988b. Growth, fedundity, and energy stores of white sucker (Catostomus commersoni) from lakes containing elevated levels of copper and zinc. Can. J. Fish. Aquat. Sci. 47: 1455-1462.

Munkittrick, K.R., D.G. Dixon. 1988c. In situ assessment of copper and zinc impacts on white sucker populations of the Manitouwadge chain of lakes. Final report, Ontario Ministry of Environment.

Munkittrick, K.R., D.G. Dixon. 1989a. Effects of natural exposure to copper and zinc on egg size and larval copper tolerance in white sucker (Catostomus commersoni). Ecotoxicology and Environmental Safety, 18: 15-26.

Munkittrick, K.R., D.G. Dixon. 1989b. A holistic approach to ecosystem health assessment using fish population characteristics. Hydrobiologia, 188/189: 123-135.

Munkittrick, K.R., D.G. Dixon. 1989c. Use of white sucker (Catostomus commersoni) populations to assess the health of aquatic ecosystems exposed to low-level contaminant stress. Can. J. Fish. Aquat. Sci., 46 (8): 1455-1462.

Munkittrick, K.R., P.A. Miller, D.R. Barton, D.G. Dixon. 1991. Altered performance of white sucker populations in the Manitouwadge chain of lakes is associated with changes in benthic macroinvertebrate communities as a result of copper and zinc contamination. Ecotoxicology and Environmental Safety, 21: 318-326.

Munkittrick, K.R. 1992. A review and evaluation of study design considerations for site-specifically assessing the health of fish populations. Journal of Aquatic Ecosystem Health, 1: 283-293.

National Water Well Association. 1989. Design and construction of water wells - A guide for engineers. Van Nostrand Reinhold, New York, NY.

National Uranium Tailings Program. 1985. Uranium tailings sampling manual. NUTP-1E. Energy Mines and Resources Canada.

Neame, P.A., J.R. Dean, B.G. Zytaruk. 1982. Distribution and concentrations of naturally-occurring radionuclides in sediments in a uranium mining area of northern Saskatchewan, Canada. Hydrobiologia, 91: 355-361.

Nelson, S.M. and R.A. Roline. 1996. Recovery of a stream macroinvertebrate community from mine drainage disturbance. Hydrobiologia 339(1-3): 73-84

Neilsen, L.A. and D.L. Johnson. 1983. Fisheries techniques. American Fisheries Society, Bethesda, Maryland.

Newbury, R.W. and M.N. Gaboury. 1993. Stream analysis and fish habitat design - a yield manual. Newbury Hydraulics, Gibsons, B.C.

Newman, M.C. 1995. Quantitative methods in aquatic ecotoxicolgy. CRC press, Lewis Publishers.

NOAA (National Oceanic and Atmospheric Administration). 1989. Standard analytical procedures of the NOAA National Analytical Facility. 2nd ed. NOAA Tech. Memo. NMFS F/NWS-092, 1986-86. Contact: National Status and Trends Program, National Oceanic and Atmospheric Administration, NOAA N/OMA32, 11400 Rockville Pike, Rockville, MD 20852.

Nolan, Davis and Associates (N.B.) Limited. 1993. Draft. Field procedures manual gas transfer measurements waste rock piles Heath Steele Mines New Brunswick. Prepared in association with Australian Nuclear Science and Technology Organisation for CANMET, File No. F92-081, May.

Nordstrum, D.K., J.M. Burchard, and C.N. Alpers. 1990. The production and seasonal variability of acid mine drainage from Iron Mountain, California: A superfund site. undergoing rehabilitation In. Designing for closure, J.W. Gadsby, J.A. Malick, and S.J. Day. (Eds.), BiTech Publ. Vancouver, B.C., pp 13-21.

Northcote, T. G., and G. L. Ennis. 1994. Mountain whitefish biology and habitat use in relation to compensation and improvement possibilities. Rev. Fish. Sci. 2:347_371.

Norris, J. 1988. Techniques for sampling surface and industrial waters: special considerations and choices. In. Principles of Environmental Sampling. Keith, L (Ed.). Washington DC, American Chemical Society. pp 247-253.

Norris, R.H. 1986. Mine waste pollution of the Molonglo River, New South Wales and the Australian Capital Territory: Effectiveness of remedial works at Captains Flat mining area. Aust. J. Mar. Freshw. Res., 37: 147-157.

Norris, R.H. and A. Georges. 1993. Analysis and interpretation of benthic macroinvertebrate surveys. In. Freshwater Biomonitoring and Benthic Macroinvertebrtaes. Rosenberg, D.M. and V.H. Resh (Eds.). pp. 234-286.

Norris, R.H., P.S. Lake, and R. Swain. 1982. Ecological effects of mine effluents on the South Esk River, North-Eastern Tasmania III. Benthic Macroinvertebrates. Aust. J. Mar. Freshw. Res., 33: 789-809.

NRC (National Research Council). 1990. Managing troubled waters: The role of marine environmental monitoring. National Academy Press. Washington, D.C.

NRCC 1985. The role of biochemical indicators in the assessment of ecosystem health - their development and validation, National research Council Canada, Associate Committee on Scientific Criteria for environmental Quality, NRCC report No. 24371.

Nuckels, T.E. 1990. Development of small diameter wells. *In*. Proceeding of the Fourth National Outdoor Action Conference on Aquifer Restoration, Groundwater Monitoring and Geophysical Methods, Water Well Journal Publishing Co., Dublin, OH.

O'Brien, R.G. and K.E. Muller. 1993. Unified power analysis for t-tests through multivariate hypotheses. In. Applied analysis of variance in behavioural science. L.E. Edwards (Ed.), New York, Marcel Dekker, pp 297-344.

OMNDM (Ontario Ministry of Northern Development and Mines). 1992. Rehabilitation Mines Guidelines for Proponents. Version 1.2.

Osborne, L.L., R.W. Davies, and K.J. Linton. 1980. Use of hierarchical diversity indices in lotic community analysis. Journal of Applied Ecology. 17: 567-580.

Oscarson, D.W., W.K. Liaw, P.M. Huang. 1981. The kinetics and components involved in the oxidation of arsenite by freshwater lake sediments. Verh. Internat. Verein. Limnol., 21: 181-186.

Palace, V.P., J.F. Klaverkamp. 1993. Variation of hepatic enzymes in three species of freshwater fish from precambrian shield lakes and the effect of cadmium exposure. Comp. Biochem. Physiol., 104C (1): 147-154.

Pardo, I. 1994. Comparative water quality characterization by PCA of an unperturbed and a polluted stream. Schweizerbart'sche Verlagsbschhandlung, D-70176 Stuttgart.

Parker, J.M., S.S.D. Foster. 1986. Groundwater monitoring for early warning of diffuse pollution. IAHS Publ. No. 157.

Pavoni, B., A. Marcomini, A. Sfriso, A.A. Orio. 1988. Multivariate analysis of heavy metal concentrations in sediments of the lagoon of Venice. The Science of the Total Environment, 77: 189-202.

Pearson, T.H., J.S. Gray, and P.J. Johannessen. 1983. Objective selection of sensitive species indicative of pollution-induced change in benthic communities.2. Data analyses. Marine ecology, 12: 237-255.

Peck, R.B. 1969. Advantages and limitations of the observational method in applied soil mechanics. Geotechnique, 19 (2): 171-187.

Peckarsky, B.L. 1984. Sampling the stream benthos. In. A manual on methods for the assessment of secondary productivity in fresh waters, 2nd Ed. Downing, J.A. and F.H. Rigler (Eds.). IBP Handbook 17. Blackwell Scientific Pubs. pp. 131-160.

Pedersen, T.F. 1984. Interstitial water metabolite chemistry in a marine mine tailings deposit, Rupert Inlet, B.C. Can. J. Earth Sci., 21: 1-9.

Pedersen, T.F. 1985. Early diagenesis of copper and molybdenum in mine tailings and natural sediments in Rupert and Holbert inlets, British Columbia. Can. J. Earth Sci., 22: 1474-1484.

Peine, A. And S. Peiffer. 1996. Neutralization processes in acid mine lake sediments. Ergebnisse der Lomnologie, 0(48): 261-267.

Pierce, C.L., J.B. Fasmussen, and W.C. Leggett. 1990. Sampling littoral fish with a seine: Corrections for variable capture effiency. Can. J. Fish. Aquat. Sci. 47: 1004-1010.

Pereira, E.G., J. Moura, J.R. Costa, J.D. Mahony, and R.V. Thomann. 1995. The S. Domingos Mine: a study of heavy metal contamination in the water column and sediments of the Chanca River basin by discharge from an ancient cupriferous pyrite mine (Portugal). Marine and Freshwater Research 46(1): 145-151.

Peterman, R.M. 1990. Statistical power analysis can improve fisheries research and management. Can. J. Fish. Aquat. Sci., 47: 2 - 15.

Petersen , R.C., and K.W. Cummins. 1974. Leaf processing in a woodland stream. Freshwater Biol. 4:343-368.

Peterson, G.S., G.T. Ankley, E.N. Leonard. 1996. Effect of bioturbation on metal-sulfide oxidation in surficial freshwater sediments. Environmental Toxicology and Chemistry, 15 (12): 2147-2155.

Pettit, C.M., M.K. Tai, and J.L. Kirkaldy. 1997. Statistical approach to geochemical assessment of waste rock. Proceedings of the Fourth International Conference on Acid Rock Drainage.

Phillips, D.J.H. and P.S. Rainbow. 1988. Barnacles and mussels as biomonitors of trace elements: a comparative study. Mar. Ecol. progr. Ser. 49: 83-93.

Phillips, D.J.H. and P.S. Rainbow. 1993. Biomonitoring of aquatic trace contaminants. Environmental Management Series. Editors, J. Cairns and R.M. Harrison. Elsevier Applied Sciences.

Plumb, R.H. Jr. 1981. Procedures for handling and chemical analysis of sediment and water sampled. Technical report. EPA/CE-81-1. United States Environmental Protection Agency and United States Army Corps of Engineers, Waterways and Experiment Station, Vicksburg, MS.

Porter, S.P. 1986. A description of measurement error near limits of detection. IAHS Publ. No. 157.

Poulin, R. 1992. Toxic pollution and parasitism in freshwater fish. Parasitology Today, 8 (2): 58-61.

Premdas, P.D. and C.D. Metcalfe 1995. Experimental transmission of epidermal lip papillomas in white sucker, Catostomas commersoni. Canadian Journal of Fisheries and Aquatic Sciences, 53: 1018-1029.

Premdas, P.D, T.L. Metcalfe, M.E. Bailey, and C.D. Metcalfe. 1995. The prevalence and hisological appearance of lip papillomas in white suckers (*Catostomus commersoni*) from two sites in central Ontario, Canada. J. Great Lakes Res. 21(2): 207-218.

Price, N.M. and F.M.M. Morel. 1994. Trace metal nutrition and toxicity in phytoplankton. Erg. Der. Limn. 42: 79-97.

Price, W.A., J. Errington, and V. Koyanagl. 1997a. Guidelines for the prediction of acidic rock drainage and metal leaching for mines in British Columbia: Part I. General Procedures and Information Requirements. Proceedings of the 4th International Conference on Acid Rock Drainage. Vol. 1: 1-14.

Price, W.A., K. Morin, and N. Hutt. 1997b. Guidelines for the prediction of acidic rock drainage and metal leaching for mines in British Columbia: Part II. General Procedures and Information Requirements. Proceedings of the 4th International Conference on Acid Rock Drainage. Vol. 1: 15-30.

Prud'homme,A.M., J. Bureau, and H.H Sloterdijk. 1993. Effect of storage conditions of white sucker liver on some MFO activity and metallothionein concentration. Science of the Total Environment 0(Suppl. Part 1): 655-661

PSEP (Puget Sound Estuary Program). 1986 (and 1989 revisions). Recommended protocols for measuring selected environmental variables in Puget Sound. Prepared by Tetra Tech, Inc. for the United States Environmental Protection Agency (USEPA), Region 10, Office of Coastal Waters, Seattle, WA.

PSEP. 1989a. Recommended guidelines for measuring selected environmental variables in Puget Sound sediment and tissue samples. Prepared by PTI Environmental Services for USEPA, Region 10, Office of Puget Sound, Seattle, WA.

PSEP. 1989b. Recommended guidelines for measuring metals in Puget Sound water, sediment, and tissue samples. Prepared by PTI Environmental Services for USEPA, Region 10, Office of Puget Sound, Seattle, WA.

PSEP. 1991. A project manager’s guide to requesting and evaluating chemical analyses. Prepared by PTI Environmental Services for Puget Sound Estuary Program, USEPA, Region 10, Office of Coastal Waters, Seattle, WA.

Qadri, S.U. 1968. Growth and reproduction of lake whitefish, *Coregonus clupeaformis*, in Lac la Ronge, Saskatchewan. J. Fish. Res. Board Can. 25:2091-2100.

Ragusa, S.R. and J.C. Madgwick. 1993. the use of autoradiography to accurately enumerate viable iron oxidizing leaching bacteria. In. Biohydrometallurgical Technologies. A.E. Torma, J.E. Wey, and V.L. Lakshmanan (Eds). The Minerals, Metals and materials Society, 433-442.

Rai, L.C., J.P. Gaur, and H.D. Kumar. 1981. Phycology and heavy metal pollution. Biol. Rev. 56: 99-151.

Raleigh, R. F. 1982. Habitat suitability index models: brook trout. FWS/OBS 82/10.24.

Raleigh, R. F., and P. C. Nelson. 1985. Habitat suitability index models and instream flow suitability curves: pink salmon. Biological Report 82(10.109).

Raleigh, R. F., T. Hickman, R. C. Solomon and P. C. Nelson. 1984a. Habitat suitability information: rainbow trout. U. S. Fish. Wildl. Serv. FWS/OBS_82/10.60. 64 pp.

Raleigh, R. F., L. D. Zuckerman, and P. C. Nelson. 1984b. Habitat suitability index models and instream flow suitability curves: brown trout. FWS/OBS 82/10.71.p.

Raleigh, R. F., W. J. Miller and P. C. Nelson. 1986. Habitat suitability index models and instream flow suitability curves: chinook salmon. U. S. Fish and Wildlife Service Biological Report 82(10.122).

Rasche, R.L. 1984. Mapping - surface or ground surveys. In: Ecological assessment of macrophyton: collection, use and meaning of data. (Eds). W.M. Dennis and B.G. Isom. Spec. Tech. Publ. 843. American Society Testing and Materials.

Rainbow, P.S. P.G. Moore and D. Watson. 1989. Talitrid amphipods as biomonitors for copper and zinc. Estur. Cstl. Shelf. Sci. 28: 245-262.

Reible, D.D., V. Popov, K.T. Valsaraj, L.J. Thibodeaux, F. Lin, M. Dikshit, M. Todaro, and J.W. Fleeger. 1996. Contaminant fluxes from sediment due to tubificid oligochaete bioturbation. Water Research 30(3): 704-714.

Reice, S.R. 1980. The role of substratum in benthic macroinvertebrate microdistribution and litter decomposition in a woodland stream. Ecology 61:580-590.

Resh, V.H. and J.K. Jackson. 1993. Rapid assessment approaches to biomonitoring using benthic macroinvertebrates. In. Freshwater Biomonitoring and Benthic Macroinvertebrtaes. Rosenberg, D.M. and V.H. Resh (Eds.). pp.195-233.

Resh, V.H. and E.P. McElravy. 1993. Contemporary quantitative approaches to biomonitoring using benthic macroinvertebrates. *In*. Freshwater Biomonitoring and Benthic Macroinvertebrtaes. Rosenberg, D.M. and V.H. Resh (Eds.). pp.159-194.

Reynolds, J. B. 1983. Electrofishing. *In*. Fisheries Techniques. L. A. Neilsen and D. L. Johnson (Eds.). American Fisheries Society. Bethesda, Maryland pp. 147-164.

Reynoldson, T.B. 1987. Interactions between sediment contaminants and benthic organisms. Hydrobiologia, 149: 53-66.

Reynoldson, T.B., K.E. Day, C. Clarke and D. Milani. 1994. Effect of indiggenous animals on chronic endpoints in freshwater sediment toxicity tests. Environ. Toxicol. Chem. 13: 973-977.

Reynoldson, T.B., R.C. Bailey, K.E. Day, R.H. Norris. 1995. Biological guidelines for freshwater sediment based on benthic assessment of sediment (the beast) using a multivariate approach for predicting biological state. Australian Journal of Ecology, 20: 198-219.

Rice, C.D., D.H. Kergosien, and S.M. Adams. 1996. Innate immune function as a bioindicator of pollution stress in fish. Ecotoxicology and Environmental Safety, 33(2): 186-192.

Richards, K. 1982. Rivers, form and process in alluvial channels. Methuen & Co., London and New York

Rieman, B. E., and J. D. McIntyre. 1993. Demographic and habitat requirements for conservation of bull trout. Gen. Tech. Rep. INT_302.Ogden, Ut:U.S. Dept. of Agric., Forest Service, Intermountain Research Station. 38p.

Ritcey, G.M. and M. Silver. 1982. Lysimeter investigations on uranium tailings at CANMET. CIM Bull., 75: 134-143.

Roark, S., K. Brown. 1996. Effects of metal contamination from mine tailings on allozyme distributions of populations of Great Plains fishes. Environmental Toxicology and Chemistry, 15 (6): 921-927.

Robertson, J.D. 1991. Subaqueous disposal of reactive mine waste: An overview of the practice with case studies. Prepared for The Second International Conference on the Abatement of Acidic Drainage.

Robertson, A.M. and J. Barton-Bridges. 1990. Cost effective methods of long term acid mine drainage control from waste rock piles. In. Designing for closure, J.W. Gadsby, J.A. Malick, and S.J. Day. (Eds.), BiTech Publ. Vancouver, B.C., pp 153-175.

Robbins, G.A. and M.M. Gemmell. 1985. Factors requiring resolution in installing vadose zone monitoring systems. Ground Water Monitoring Review. 5: 75-80.

Robohm, R.A. 1986. Paradoxical effects of cadmium exposure on antibacterial antibody responeses in two fish specoies: inhibition in cunners (*Tautogolabrus adspersus*) and enhancement in striped basss (*Morone saxatilis*). Vet. Immunol. Immunopathol. 12: 251-256.

Rocchini, R.J., M.J.R. Clark, A.J. Jordan, S. Horvath, D.J. McLeay, J.A. Servizi, A. Sholund, H.J. Singleton, R.G. Watts, and R.H. Young. 1982. Provincial guidelines and laboratory procedures for measuring acute lethal toxicity of liquid effluents to fish. B.C. Ministry of the Environment, Victoria, B.C.

Roesijadi, G. 1992. Metallothioneins in metal regulation and toxicity in aquatic animals. Aquat. toxicol. 22: 81-114.

Rosa, F and J.M. Azcue. 1993. Peeper methodology - a detailed procedure from field experience, Canadian Centre for Inland Waters Report. 93-33.

Rosassen, A.R. 1997. Impact of minewater effluent on phytoplankton in a series of northern Saskatchewan lakes. University of Saskatchewam, MSc. Thesis.

Rosenberg, D.M. and V.H. Resh. 1982. The use of artificail substrates in the study of freshwater benthic macroinvertebrates. In. Artificail Substrates. Cairns, J. (Ed.). Ann Arbor Science Pubs. Ann Arbor Michigan. pp. 175-235.

Rosenberg, D.M. and V.H. Resh. 1993. Freshwater Biomonitoring and Benthic Macroinvertebrates Chapman & Hall Inc. New York.

Rosenberg, D.M. and A.P. Weins. 1976. Community and species responses of Chironomidae (Diptera) to contamination of fresh waters with crude oil and petroleum products, with special reference to the Trail River, Northwest territories. Journal of the Fisheries Research Board of Canada. 33: 1955-1963.

Rowan, D.J., J.B. Rasmussen, and J. Kalff. 1995. Optimal allocation of sampling effort in lake sediment studies. Canadian Journal of Fisheries and Aquatic Sciences, 52; 2146-2158.

Sakai, H., Y. Kojima, and K. Saito. 1986. Distribution of heavy metals in water and seived sediments in the Toyohira River. Water Res. 20(5): 559-567.

Salomons, W., N.M. de Rooij, H. Kerdijk, and J. Bril. Sediments as a source for contaminants. In. Ecological Effects of In-Situ Contaminants. Evans, R.T.R., A. Hamilton, M. Munawar, T. Reynoldson, and H. Sadar (eds.), Hydrobiologia 149: 13-30.

Schafer, W.M. 1992. Acid forming mining waste: Prediction, control and treatment. In: Randol, Vancouver, 92, pp. 345-353.

Scharer, J.M. and R.V. Nicholson. 1991. Coupled processes to extrapolate the kinetics of pyrrhotite and pyrite oxidation from the laboratory to mine tailings in the field. Proceedings of the 1991 Geological Society of America Meeting, San Diego, California, U.S. A., August.

Schippers, A., R. Hallman, S. Wentzien, and W. Sand. 1995. Microbial diversity in uranium mine heaps. Applied and Environmental Microbiology, 61(8): 2930-2935.

Schroeder, P., R. Peyton, B. McEnroe, and J. Simpson. 1989. The hydrologic evaluation of landfill performancve (HELP) Model, user's guide for Version 2. U.S. Army Engineer Waterways Experiment Staiton, Vicksburg, Miss. Interagency Agreement No. DW21931425-01-3.

Schubert, L.E. 1984. Algae as ecological indicators. Academic Press. London.

Schults, D.W., S.P. Ferrarro, L.M. Smith, F.A. Roberts and C.K. Poindexter. 1992. A comparison of methods for collecting interstitial water for trace organic compounds and metal analyses. Wat. Res. 26: 989-995.

SEPS. 1992. Mine rock guidelines design and control of drainage water quality. Report 93301. Prepared by Steffen, Robertson, Kirsten (Canada) Inc.

Shaw, E.M. 1983. Hydrology in practice. Van Nostrand Rienhold (UK) Co. Ltd., Berkshire.

Shaw, J.L., J.P. Manning. 1996. Evaluating macroinvertebrate population and community level effects in outdoor microcosms: Use of in situ bioassays and multivariate analysis. Environmental Toxicology and Chemistry, 15 (5): 608-617.

Sibley, P.K., G.T. Ankley, A.M. Cotter, E.N. Leonard. 1996. Predicting chronic toxicity of sediments spiked with zinc: An evaluation of the acid-volatile sulfide model using a life-cycle test with the midge *Chironomus tentans*. Environmental Toxicology and Chemistry, 15 (12): 2102-2112.

Skalaski, J.R. and D.H. McKenzie. 1982. A design for aquatic monitoring programs. J. Envir. Mgmt., 14: 237-251.

Sloof, W. 1982. Skeletal anomolies in fish from polluted surface waters. aquatic Toxicology (Amsterdam), 2: 157-173.

Smeach, S.C., R.W. Jernigan. 1977. Further aspects of a markovian sampling policy for water quality monitoring. Biometrics, 33: 41-46.

Smith, W. 1984. Design of efficient environmental surveys over time. Statistics in the Environmental Sciences, ASTM STP 845.

Smith, I.R., H.W. Ferguson, and M.A. Hayes. 1989. Histopathology and prevalence of epidermal papillomas epidemic in brown bullhead, *Ictalurus nebulosus* (Lesueur), and white sucker, *Catostomus commersoni* (Lacépède), populations from Ontario, Canada.

Smith, K.S. and D.L. Macalady. 1991. Water/sediment partitioning of trace elements in a stream receiving acid-mine drainage. In. Second International Conference on the Abatement of Acidic Drainage, Tome 3, Montreal, MEND (Ed.) pp. 138-450.

Snedecor, G.W., W.G. Cochran. 1989. Statistical methods, 8th ed. Iowa State University Press, Ames, IA.

Sobek, A.A., W.A. Schuller, J.R. Freeman, R.M. Smith. 1978. Field and laboratory methods applicable to overburdens and mine-soils. United States Environmental Protection Agency, EPA-600/2-78-54, Cincinnati, OH.

Sokal, R.R., F.J. Rohlf. 1981. Biometry: The principles and practice of statistics in biological research. 2nd ed. W.H. Freeman and Co., New York, NY.

Sopuck, V.J., D.A.W. Lehto, B.T. Schreiner, and J.W.J. Smith. 1980. Interpretation of reconnaissance lake sediment data in the precambrian shield area, Saskatchewan. Saskatchewan Research Council, G78-10a.

Somlyody, L., J. Pinter, L. Koncsos, I. Hanacsek. 1986. Estimating averages and detecting trends in water quality data. IAHS Publ. No. 157.

SRK (Steffen, Robertson, and Kirsten). 1992. ARD prediction in the north. Prepared for the Department of Indian and Northern Affairs, Ottawa, ON.

SRK. 1993. Rock pile water quality modelling - Phase I final report. Prepared for the B.C. Ministry of Energy, Mines and Petroleum Resources, Victoria, BC.

St. Louis, V.L., L. Breebart, J.C. Barlow, and J.F. Klaverkamp. 1993. Metal accumulation and metallothionein concentrations in tree swallow nestlings near acidified lakes. Environ. Toxicol. and Chem. 12: 1203-1207.

Steele, T.D. 1986. Converting water quality information goals into statistical design criteria. IAHS Publ. No. 157.

Stewart-Oaten, A., W.W. Murdoch, and K.R. Parker. 1986. Environmental impact assessment: pseudoreplication in time?. Ecology 67: 929-940.

Steinman, A.D. 1993. Research on disturbance in artificial streams. Journal of the North American Benthological Society, 12 (4): 347-350.

Stokes, P.M. 1974. Uptake and accumulation of copper and nickel by metal-tolerant strains of *Scedesmus*. Verh. Internat. Wer. Limnol. 19: 2128-2137.

Stokes, P.M. 1983. Response of freshwater algae to metals. Progr. Phycol. Res. 2: 88-112.

Stokes, P.M., T.C. Hutchison, and K. Krauter. 1973. Heavy metal tolerance in algae isolated from polluted lakes near Sudbury, Ontario. Water pollut. Res. Can. 8: 178-201

Storey, A.W. and L.C. Pinder. 1985. Mesh size and efficiency of sampling of larval Chironomidae. Hydrobiologia 124: 193-197.

Stromberg, B., L. Moreno, and J. Crawford. 1995. Modelling of transport and initial weathering processes in a sulphidic mining waste rock heap. Acta Universitatis Carolinae Geologica 39(1): 171-180.

Stucky, D.J., J.H. Bauer, T.C. Lindsey. 1980. Restoration of acidic mine spoils with sewage sludge: I revegetation. Reclamation Review, 3: 129-139.

Stumm, W., J.J. Morgan. 1981. Aquatic chemistry: An introduction emphasising chemical equilibria in natural waters. John Wiley & Sons, Toronto, ON.

Suedel, B.C., E. Deaver, and J.H. Rodgers. 1996. Formulated sediment as a reference and dilution sediment in definitive toxicity tests. Arch. Environ. Contam. Toxicol. 30: 47-52.

Suter, G.W. II., J.M. Loar. 1992. Weighing the ecological risk of hazardous waste sites. Environ. Sci. Technol., 26 (3): 432-438.

Suter, G.W. II. 1993. Ecological risk assessment. Lewis Publishers, Chelsea, MI.

Suter, G.W. II. 1996. Toxicological benchmarks for screening contaminants of potential concern for effects of freshwater biota. Environmental Toxicology and Chemistry, 15 (7): 1232-1241.

Suter, G.W., A.E. Rosen, E. Linder, and D.F. Parkhurst. Endpoints for respojnses of fish to chronic toxic exposures. Environ. Toxicol. Chem. 6: 793-809.

Swift, M.C., N.H. Troelstrup, Jr., N.E. Detenbeck, J.L. Foley. 1993. Large artificial streams in toxicological and ecological research. Journal of the North American Benthological Society, 12 (4): 359-366.

Tabachnik, B.G., L.S. Fidell. 1983. Using multivariate statistics. Harper & Row, New York, NY.

TAEM 1996. Cameco Corporation Rabbit Lake uranium mine status of the environment report. Prepared for Cameco Corporation, Saskatoon, SK. Prepared by TAEM, Saskatoon.

Telford, W.M., L.P. Geldart, R.E. Sheriff, and D.A. Keys. 1976. Applied geophysics. Cambridge University Press.

ter Braak, C.J.F. and P.F.M. Verdonschot. 1995. Canonical correspondence analysis and related multivariate methods in aquatic ecology. Aquatic Sciences, 57(3): 255-289.

Terzaghi, K., R.B. Peck. 1948. Soil mechanics in engineering practice. 2nd ed. Wiley, New York, NY.

Terzi, R.A. 1981. Hydrometric field manual, measurements of streamflow. Environment Canada, Inland Water Directorate, Water Resource Branch. Ottawa, Canada.

Tessier, A., and P.G.C. Campbell. 1987. Partitioning of trace metals in sediments: Relationships with bioavailabilty. Hydrobiologia, 149: 43-52.

Tessier, A., P.G.C. Campbell, and Mason. 1984. Relationships between the partitioning of trace metals in sediments and their accumulation in the tissues of the freswater mollusc *Elliptio complanata* in a mining area. Canadian Journal of Fisheries and Aquatic Sciences. 41: 1463-1472.

Tessier, A., P.G.C. Campbell, and M. Bisson. 1979. Sequential extraction procedure for the speciation of particlulate trace metals. Analytical Chemistry 51: 844-851.

Tessier, A., P.G.C. Campbell, and M. Bisson. 1982. Particulate trace metal speciation in stream sediments and relationships with grain size: implications for geochemical exploration. Journal of Geochemical Exploration. 16: 77-104.

Tetra Tech. 1986a. Analytical methods for U.S. EPA priority pollutants and 301 (h) pesticides in estuarine and marine sediments. Final report prepared by Tetra Tech, Inc., for the U.S. Environmental Protection Agency under Contract No. 68-01-6938.

Tetra Tech. 1986b. Bioaccumulation monitoring guidance: 4. Analytical methods for U.S. EPA priority pollutants and 301 (h) pesticides in tissues from estuarine and marine organisms. Final report prepared by Tetra Tech, Inc., for the U.S. Environmental Protection agency under Contract No. 68-01-6938.

Tetra Tech. 1986c. Recommended protocols for measuring organic compounds in Puget Sound sediment and tissue samples. A report prepared for Office of Puget Sound Region 10, United States Environmental Protection Agency, Seattle, Washington.

Tetra Tech. 1986d. Bioaccumulation monitoring guidance: 5. Strategies for sample replication and compositing. Final report tetra Tech Inc. Bellevue Washington. dricoll

Thomas, P. 1990. Molecular and biochemical responses of fish to stresors and their potential use in environmental monitoring. American Fisheries Society Symposium 8, 9-28.

Thomson, E.A., S.N. Luoma, C.E. Johansson, D.J. Cain. 1984. Comparison of sediment and organisms in identifying sources of biologically available trace metal contamination. Water Res. 18 (6): 755-765.

Thompson, M. 1986. The future role of inductively-coupled plasma atomic emission spectroscopy in applied geochemistry. In: Thorton, I. and Howarth, R.J. Eds, Applied Geochemistry in the 1980s, John Wiley and Sons, pp, 191-211.

Thompson, J.M. 1992. Visual representation of data including graphical exploratory data analysis. In: Methods of Environmental Data Analysis. C.N. Hewitt. (Ed.). Elsevier Science Publishers Ltd. pp. 213-258.

Thompson, K.R. and D.W. Beckman. 1995. Validation of age estimates from white sucker otoliths. Transactions of the American Fisheries Society 124 (4): 637-639.

Thompson, P.L. and F. Bärlocher. 1989. Effect of pH on leaf breakdownn in streams and in the laboratory. journal of the North American Benthological Society, 8: 203-210.

Thorne, L. and G. Nickless. 1981. The relation between heavy metals and particle size fractions within the Severn estuary (U.K.) inter-tidal sediments. The Science of the Total Environment, 19: 207-213.

Thorstenson, D.C. 1984. The concept of electron activity and its relation to redox potentials in aqueous geochemical systems. U.S. geological Survey, Open-file 84-072.

Tibble, P.A. and R.V. Nicholson. 1997. Oxygen consumption on sulphide tailings and tailings covers: Measured rates and applications. Proceedings of the Fourth International Conference on Acid Rock Drainage.

Todd, D.K. 1959. Groundwater hydrology. John Wiley & Sons, New York, NY.

Trial, J. G., J. G. Stanley, M. Batcheller, G. Bebhart, O. E. maughan, and P. C. Nelson. 1983a. Habitat suitability index models: blacknose dace. FWS/OBS 82/10.41.

Trial, J. G., C. S. Wade, J. G. Stanley, and P. C. Nelson. 1983b. Habitat suitability index models: common shiner. FWS/OBS 82/10.40.

Trippel, F.A., H.H. Harvey. 1991. Comparison of methods used to estimate age and length of fishes at sexual maturity using populations of white sucker (*Catostomus commersoni*). Can. J. Fish Aquat. Sci., 48: 1446-1459.

Twomey, K.A., K.L. Williamson, and P.C Nelson. 1984. Habitat suitability index models and instream flow suitability curves: White sucker. U.S. Fish Wildl. Serv. FWS/OBS-82/10.64.

Underwood, A.J. 1991. Beyond BACI: Experimental designs for detecting human environmental impacts on temporal variations in natural populations. Aust. J. Mar. Freshwater Res., 42: 569-587.

Underwood, A.J. 1992. Beyond BACI: The detection of environmental impacts on populations in the real, but variable, world. J. Exp. Mar. Biol. Ecol., 161: 145-178.

Underwood, A.J. 1994. On beyond BACI: Sampling designs that might reliably detect environmental disturbances. Ecological Applications, 4 (1): 3-15.

U.S. EPA. 1978. Field and laboratory methods applicable to overburden and mine-soils. United States Environmental Protection Agency. EPA-600/2-78-054.

U.S. EPA. 1980. Guidelines and specifications for preparing quality assurance project plans. QAMS-005/80. U.S. Environmental Protection Agency, Office of Monitoring and Quality Assurance, Office of Research and Development, Washington, DC.

U.S. EPA. 1985a. Practical guide to groundwater sampling. EPA-600/2-858104. United States Environmental Protection Agency, Environmental Research Laboratory, Ada, OK.

U.S. EPA. 1985b. Methods manual for bottom sediment sample collection. EPA-905/4-85-004. USEPA, Great Lakes National Program Office, Chicago, IL.

U.S. EPA. 1986a. Development of data quality objectives. Description of stages I and II. Prepared by Quality Assurance Management Staff, U.S. Environmental Protection Agency, Office of Research and Development, Washington, DC.

U.S. EPA. 1986b. Test methods for evaluating solid waste (SW-846): Physical/chemical methods. USEPA, Office of Solid Waste, Washington, DC.

U.S. EPA 1987a Guidance for sampling of and analyzing for organic contaminants in sediments. Office of Water, Regulations, and Standards Division, Washington, D.C.

U.S. EPA. 1987a. Quality assurance/quality control (QA/QC) for 301 (h) monitoring programs: Guidance on field and laboratory methods. EPA 430/9-86-004. Prepared by Tetra Tech, Inc., Bellevue WA. For the U.S. Environmental Protection Agency Office of Marine and Estuarine Protection. NTIS Number PB87-221164.

U.S. EPA. 1987b. Handbook: Groundwater. U.S. Environmental Protection Agency, Center for Environmental Research Information, Cincinnati, OH.

U.S. EPA. 1988. MINTEQ. Centre for Exposure Assessment Modeling, U.S. Environmental Protection Agency.

U.S. EPA. 1989a. Groundwater sampling for metals analyses. EPA-540/4-89-001. United States Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC.

U.S. EPA. 1989b. Quality assurance handbook for air pollution measuremenst systems. Volume IV - Meteorological measurements. Office of Research and Development, Atmospheric research and Exposure Assessment Laboratory, North Carolina.

U.S. EPA. 1989c. Short-term methods for estimating chronic toxicity of effluents and receiving waters to freshwater organisms. 2nd Ed. Report EPA/600/4-89/001. United States Environmental Protection Agency.

U.S. EPA. 1991a. A project manager's guide to requesting and evaluating chemical analyses. Prepared by PTI Environmental Services for the United States Environmental Protection Agency, Region, 10, Office of Coastal Waters, Seattle, WA.

U.S. EPA. 1991b. Handbook, ground water volume: II: methodology. U.S. Environmental Protection Agency, Office of Research and Development. EPA/625/6-90/016b. Center for Environmental Research Information.

U.S. EPA. 1993. Methods for measuring the acute toxicity of effluents and receiving waters to freshwater and marine organisms, Report EPA/600/4-90/027. United States Environmental Protection Agency, Cincinnati, OH.

U.S. EPA. 1994. Short-term methods for estimating the chronic toxicity of effluents and receiving waters to freshwater organisms, 3rd Ed. Report EPA/600/4-91-002. United States Environmental Protection Agency, Cincinnati, OH.

US NSC (United States National Security Council). 1996. Fact sheet - U.S. global positioning system policy. March 29, 1996, Press Release, Office of Science and Technology Policy.

Valiela, D., P.H. Whitfield. 1989. Monitoring strategies to determine compliance with objectives. Water Resources Bulletin, 25: 63-69.

Vanriel, P. and L. Johnson. 1996. Action principles as determinants of ecosystem structure: the autonomous lake as a reference system. Ecology 76(6): 1741-1757.

van Zyl, D. 1987. Health risk assessment and geotechnical perspective. In: Geotechnical Practice for Waste Disposal. (Ed) R.D. Wood. ASCE. pp. 812-831.

Vinyard, G.L. 1996. A chemical and biological assessment of water quality impacts from acid mine drainage in a first order mountain stream, and a comparison of two bioassay techniques. Environmental Technology 17(3): 273-281.

Voellestad, L. A., J. Skurdal and T. Qvenild. 1986. Habitat use, growth, and feeding of pike (*Esox lucius*) in four Norwegian lakes. Arch. Hydrobiol. 108:107-117.

Wallis, P.M. and T.I Ladd. 1984. Review of microbial processes on the evolution of groundwater chemistry with some practical applications for coal mine reclamation studies in Alberta. In. Proceeding of the first Canadian/American conference on hydrogeology, Banff Alberta, Eds, B. Hitchon and E.G. Wallick. National Water Well Association.

Wald, A. 1947. Sequential analysis. John Wiley, New York, NY.

Ward, R.C. 1986. Framework for designing water quality information systems. IAHS Publ. No. 157.

Ward, R.C., J.C. Loftis, and G.B. McBride. 1986. The data rich but information poor syndrome in water quality monitoring. Environmental Management 10(3): 291-297.

Washington, H.G. 1984. Diversity, biotic and similarity indices. A review with special relevance to aquatic ecosystems. Water Research 18: 653-694.

Waterhouse, J.C., M.P. Farrell. 1985. Identifying pollution related changes in chironomid communities as a function of taxonomic rank. Can. J. Fish. Aquat. Sci., 42: 406-413.

Wedemeyer, G.A. and D.J. McLeay. 1981. Methods for determining the tolerance of fishes to environmental stressors. *In*. Stress and Fish. A.D. Pickering (Ed.). pp. 247-275.

Weeks, E.P. 1978. Field determination of vertical permeability to air in the unsaturated zone. U.S. Geol. Surv. Prof. Paper, 1051:1-41.

Weeks, J.M. and P.G. Moore. 1991. The effects of synchronous molting on body copper and zinc concentrations in four species of talitrid amphipods (Crustacea). J. mar. Biol. Ass. U.K. 71: 481-488.

Wege, G.J. and R.O. Anderson. 1978. Relative weight ($W_r$) a new index of condition for largemouth bass. In. New Approaches to the Management of Small Impoundments. Novinger, G.D. and J.G. Dillard (Eds.). American Fisheries Society, Special Publication 5, Bethesda, Maryland.

Weis, P. and J.S. Weis. 1991. The developmental toxicity of metals and metalloids in fish. In. Metal Ecotoxicology: Concepts and Applications. A.W. McIntosh (Ed.). Lewis publ. Chelsea, Michigan. pp. 145-169.

Wester, P.W., A.D. Vethaak, and W.B. van Muiswinkel. 1994. Fish as biomarkers in immunotoxicology. Toxicology, 86: 213-232.

Wetzel, R.G. 1983. Limnology. CBS College Publishing, Philadelphia.

Wetzel, R.G. and G.E. Likens. 1990. Limnological analyses 2nd Ed. Springer-Verlag. London.

Whittaker, R.H. 1978. Ordination of Plant Communities. W. Junk, The Hague.

Whitfield, P.H. and P.F. Woods. 1984. Intervention analysis of water quality records. Water Resources Bulletin, 20: 657-668.

Whitfield, P.H. 1988. Goals and data collection designs for water quality monitoring. Water Resources Bulletin, 19: 115-121.

Whitton, B.A. 1984. Algae as indicators of heavy metals in freshwaters. In. Algae as Ecological Indicators. Schubert, L.E. (Ed.). Academic Press. London. pp. 257-280.

Wiederholm, T. 1984. Responses of aquatic insects to environmental pollution. In. The Ecology of Aquatic Insects. Resh, V.H. and D.M. Rosenberg (Eds.] Praeger Publishers, New York, NY. pp. 508-557.

Wilkinson, L. 1990. SYSTAT: The system for statistics. SYSTAT, Inc., Evanston, Illinois, pp. 677.

Wilson, L.G., L.G. Everett, and S.J. Cullen. 1995. Handbook of vadose zone characterization and monitoring. Lewis Publishers London..

Winer, B.J. 1971. Statistical principles in experimental design. 2nd ed. McGraw-Hill, New York, NY.

Winner, R.W., B.W. Boesel, and M.P. Farrell. 1980. Insect community structure as an index of heavy-metal pollution in lotic ecosystems. Can. J. Fish. Aquat. Sci. 37: 647-655.

Wolf 1988. Fish viruses and fish viral diseases. Cornell Univ. Press. Ithica, New York.

Wong, C.S., F.A Whitney, W.K. Johnson. 1986. The effect of mine tailings on levels of heavy metals in seawater. ACTA Oceanologica SINICA, 5 (3): 401-415.

WQNL (Water Quality National Laboratory). 1985. Protocol on preservation container type and bottle preperation, Water Quality National Laboratory, Burlington, Canada.

Yan, N.D. 1979. Phytoplankton community of an acidified, heavy metal-contaminated lake near Sudbury, Ontario. Water, Air and Soil Pollution, 11(1): 43-55.

Yan, N.D. and R. Strus. 1980. Crustacean zooplankton communitites of acidic, metal-contaminated lakes near Sudbury, Ontario. Canadian Journal of Fisheries and Aquatic Sciences, 37(12): 2282-2291.

Yanful, E.K. and L.C. Arnaud. 1991. The Waite Amulet hydrogeochemical evaluation. In. Second International Conference on the Abatement of Acidic Drainage, Tome 1, Montreal, MEND (Ed.) pp. 91-114.

Young, P.C., T. Young. 1992. Environmetric methods of nonstationary time-series analysis: Univariate methods. In: Methods of Environmental Data Analysis, (ed) C.N. Hewitt. Elsevier Science Publishers Ltd.

Zauke, G.P. 1982. Cadmium in Gammaridae (Amphipoda: Crustacea) of the Rivers Werra and Weser - II. Seasonal variation and correlation to temperature and other environmental variables. Water Res. 16: 785-792.

# LIST OF TABLES

## CHAPTER I

## CHAPTER II

## CHAPTER III

## CHAPTER IV

## LIST OF FIGURES

### CHAPTER I

### CHAPTER II

### CHAPTER III

**CHAPTER IV**

## APPENDIX I – TECHNICAL SUMMARY NOTES *(separate document)*

## TABLE II: 1.1-1

Summary of common sulphide minerals and their oxidation products (from BC AMD 1989a).

| 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 | $Fe_{1-x}S$ | $Fe^{3+}$, $SO_4^{2-} \cdot H^+$ | 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 | $Fe^{3+}$, $SO4^{2-}$, $H^+$ | 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 |
| Bomite | $Cu_5FeS_4$ | $Cu^{2+}$, $Fe^{3+}$, $SO_4^{2-}$, $H^+$ | Ferric hydroxides and sulphates; copper hydroxides and carbonates; gypsum |
| Arsenopyrite | FeAsS | $Fe^{3+}$, $AsO_4^{3-}$, $H^+$ | 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 |
| Molybdenjte | $MoS_2$ | $MoO_4^{2-}$, $SO_4^{2-}$, 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 | Cd+, $AsO_4^{3-}$, $SO_4^{2-}$, $H^+$ | 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.

# TABLE II: 1.1-3

Summary of acid consuming minerals and their neutralizing characteristics (from BC AMD 1989a).

| **Mineral** | **Composition** | **Acid Consuming Potential**[a] | **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 | $FeO(OH)$ | 89 | 3.0 - 3.7 |
| Manganite | $MnO(OH)$ | 88 | |
| Brucite | $Mg(OH)_2$ | 29 | |

[a] Acid consuming potential is given as the weight (g) of the mineral required to have the same neutralizing 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).

# TABLE II: 1.3-1

Ranking of common ore deposit types in Western Canada according to AMD potential’.

| Mined Deposit Type | Examples | ARD Susceptibility |
|---|---|---|
| Sediientary massive sulphides (SEDEX) | Sullivan, Tom (Yukon) | Most prone |
| Volcanogenic massive sulphides (VMS) | | |
| a) Kuroko-type | Westmin, Britannia | |
| b) Besshi-type | Windy Craggy, Ruttan | |
| c) Cyprus-type | Chu Chua, Anyox | |
| Calc-alkaline suite porphyry deposits | Gibraltar, Island Copper | ↕ |
| Epithermal Au-Ag deposits | Cinnola, Premier Gold | |
| Mesothermal vein Au deposits | Snip, Frasergold | |
| Alkaline suite porphyry deposits | Afton, Copper Mountain | |
| Skarns | Craigmont, Nickel Plate | Least prone |

[1] From MEND 1.32.1

# TABLE II: 1.1-2

Bacterial species which influence rate of sulphur and iron oxidation (from BC AMD 1989a).

| Bacteria Species | Type | Optimal Growth Chemical Environment | Reference |
|---|---|---|---|
| *Thiobacillus ferro-oxidans* | Sulphur oxidizing<br>Iron oxidizing | pH = 2.5 - 3.5 | |
| *T. novellus* | Sulphur oxidizing | pH = neutral to alkaline | |
| *T. thioporus* | Sulphur oxidizing | | |
| *T. denitiricans* | Sulphur oxidizing | pH = neutral to alkaline<br>Nitrate supply for reduction to $N_2$ | |
| *Arthrobacter* sp. | Sulphur oxidizing | | Wallis and Ladd (1984 |
| *Bacillus* sp. | Sulphur oxidizing | | |
| *Flavobacterium* sp. | Sulphur oxidizing | | |
| *Pseudomonas* sp. | Suiphur oxidizing | | |
| *Desulfavibrio* sp. | Sulphur reducing | | Wallis and Ladd (1984 |
| *Desulfotomaculum sp.* | Sulphur reducing | | |
| *Salmonells* sp. | Suiphur reducing | | |
| *Proteus* sp. | Sulphur reducing | | |
| *Suffolosusp. sic* | Iron oxidizing | | Wallis and Ladd (1984 |
| *Metalloginium* sp. | Iron oxidizing | | |
| *Sidenocapsa* sp. | Iron oxidizing | | |
| *Leptothrax* sp. | Iron oxidizing | | |
| *Gallionella* sp. | Iron oxidizing | | |
| *Vibrio* sp., *Bacillus* sp. | Iron reducing | | |
| *Aerobacter aerogenus* | Iron reducing | | |

## TABLE II: 2.6-1

Quality assurance options for laboratory analyses.

| **Laboratory-Initiated Quality Assurance Options** | |
|---|---|
| **Test** | **Description** |
| Method Blank | Usually distilled water with added reagents, which is carried through the entire analysis as a check on laboratory contamination (also called a reagent blank). At least one full method blank should be run for each batch. |
| Duplicate | A homogenous sample is split either in the field (field duplicate) or in the laboratory prior to digestion (analytical duplicate) with the duplicate presented to the analyst as an additional sample to check for precision. Duplicates should not be tested consecutively. At least 15-20% or one sample per-batch (whichever is greater) should be duplicated to provide an indication of reproducibility. If more than two splits are analyzed the term replicate is normally used. |
| Check Standard | A procedure that is standardized with calibration standards prior to analyzing the samples. The analytical response to the standards is checked by frequently analyzing one or more standards along with samples. The check standards are prepared independently of the calibration standards. |
| Spike | A known amount of analyte added to a sample to provide information on matrix effects (on compounds of interest) and apparent accuracy. Surrogate spike compounds can be used to evaluate analytical recovery from each sample. |
| Standard Reference ("Control") Material | A material that contains a known concentration of the analyte in question. Based upon a reliable documentation of the analyte concentration, a reference material is certified by agencies such as the NIST and the; standard reference materials are available from NIST and CCRP. At least one certified reference material should be analyzed per batch. For those parameters where standard reference materials are available, these can be analyzed in lieu of parameter spikes. |

## TABLE II: 2.6-1

Quality assurance options for laboratory analyses.

| Field-Initiated Quality Assurance for Analytical Tests | |
|---|---|
| **Test** | **Description** |
| Transfer (Preservation) Blanks | A sample container is filled with distilled water to the same volume as that for samples and preserved as if it were a normal water or sediment sample. This blank is then sent to the laboratory for analysis. |
| Cross-Contamination Blanks | Decontaminated sample-handling equipment (spatulas, augers, core barrels) are wiped with a clean lab tissue, which is then placed in a sealable container. Alternatively, equipment is rinsed with distilled-deionized water, and the water is collected and preserved as if it were a normal sample. |
| Blind Replicate Samples | Collected sample is homogenized and split in the field into at least three identical aliquots, and each aliquot is treated and identified as a separate sample. The replicates are sent blind to the laboratory. The mean, standard deviation, and relative percent standard deviation are calculated by the project QA/QC coordinator. Alternatively/in addition, a collected sample may be split in the field into two aliquots, and one aliquot sent for analysis to a different or "reference" laboratory. The relative percent difference is calculated by the project QA/QC coordinator. If project constraints require the use of more than one laboratory, their comparability must be established using certified reference materials. |
| Blind Standard Reference Materials | Standard reference material is placed in a sample container at the time of sample collection and sent blind to the laboratory. The percent recovery is calculated by the project QA/QC coordinator. |

## TABLE III: 1.1-1

Techniques for assessing AMD potential.

| Test Classification | Description |
|---|---|
| Previous Experience or Geographical Comparisons | Examination of predictive test records and/or sampling of a drainage from abandoned or operating mines near the project study area. Use experience gained from other projects with similar geological characteristics. |
| **Paleoenvironmental and Geological Models** | Paleoenvironmental models can be used to examine the regional depositional environment of sulphide minerals contained in the component under study. Geological models are valuable in estimating the location and size of potentially acid generating zones. |
| **Chemical, Mineralogical and Physical Analysis of the Waste** | Chemical, mineralogical, and physical analysis of waste components are carried out usually as part of preliminary characterization of the material, often in **combination with the two previously mentioned techniques. These analyses can also provide a useful and important** means of extending the scope of detailed static and kinetic prediction test programs. |
| **Identifying Extractable Metals** | **Although not usually included in ARD testing protocol, tests to determine readily extractable metals are often advised to provide data** on the short term leaching **characteristics of waste components and as preliminary data prior to kinetic** |
| **Static Tests** | **Static prediction tests are simple tests to compare the balance between the acid generating components (sulphides) and the acid consuming components (principally carbonates) of the waste.** |
| Kinetic Tests | **Kinetic prediction test attempt to assess, over time, the acid producing and** consuming processes, including a prediction of drainage quality, in the laboratory |
| Mathematical Models | Predictive modelling of AND potential is a recent development. The objective is to quantitatively predict mine drainage quality that will occur beyond the time framework of the laboratory or field kinetic test. These models are complex and still require verification. |

## TABLE III:1.2-1

Characteristics of static AMD tests[1].

| **Test Type and References** | **Advantages** | **Disadvantages** |
|---|---|---|
| Fizz Test<br>Coastech 1991<br>Horberger 1989 | -easy to apply<br>-can be used in the field<br>-indicator of carbonate | -qualitative only<br>-cannot be applied for prediction |
| **Paste pH**<br>Kenton et al. 1989 | -easy to apply<br>-can be used in the field<br>-indicates net free acidity/alkalinity | -does not measure total acidity<br>-false positive+D25/negative response<br>-cannot be applied for prediction |
| **Neutralization Potential (NP)**<br>Sobek 1978<br>Coastech 1991<br>**U.S. EPA and Modified U.S. EPA Methods**<br>Norecol 1991<br>**B.C. Research Initial and Modified (NP) Tests**<br>SRK, Norecol and Gormely 1990<br>Duncan and Bruynesteyn 1979 | -well known, popular laboratory procedures<br>-quantitative measure of total buffering capacity<br>-low cost<br>-reproducible if performed properly<br>-recommended by government agencies | -gives little indication of ARD potential<br>-no mineralogy (source of neutralization potential) |
| **Alkaline Production Potential Sulphur Ratio or (APP/S)**<br>Coastech 1989 | -simple calculations<br>-rapid indicator of potential ARD<br>-useful preliminary analysis | -theoretical analysis<br>-results need to be **confirmed** by experimentation |
| **Net Acid Production (NAP)**<br>Coastech 1989<br>**Standard and Modified Acid Base Accounting**<br>Coastech 1991 | -combination of acid production and acid neutralization procedures<br>-can differentiate between sulphide minerals **(pyrrhotite/pyrite)**<br>-allows determination of carbonate and non-carbonate buffering capacity | -long, often complex procedure<br>-interpretation may be **difficult**<br>-no indication of reaction rate |
| **Net Acid Generation (NAG)**<br>Miller et al. 1990<br>**Hydrogen** Peroxide<br>Finkleman and Giffen 1990 | -simple, straight forward test procedures<br>-good reproducibility | -overestimates net acid production<br>-requires pulverized (unrepresentative) samples<br>-seldom used in Canada |
| **B.C. Research Confirmation Test[2]** | -used to confirm results of static prediction tests | -usually grouped with dynamic test methods |

[1] Adapted from MEND 1.16.1a 1994

[2] Not considered a “true” static test. Designed to confirm the results of static prediction methods (MEND 1994).

## Table III: 1.2-2

## CHARACTERISTICS OF THE DYNAMIC AMD TESTS

| Test Type | Advantages | Disadvantages |
|---|---|---|
| ***Hunidity Cells***<br><br>Caruccio et al. 1981<br>Lawrence 1990<br>Sobek et al. 1978<br>BC AMD 1989a | - Simple to set up and operate<br>- Models wet/dry cycles of the environment<br>- Rates and temporal variation in acid generation and sulphide oxidation can be determined.<br>- Effects of bacterial activity can be assessed<br>- Can be used to assess AMD control option such as blending and submerged deposition<br>- Accepted method in both Canada and the U.S.A. | - A large number of cycles are often required to complete the test due to time required to deplete NP to levels at which acid generation commences. |
| ***Column / Lysimeter Tests***<br><br>Ritcey 1989<br>Cauccio et al. 1981<br>BC AMD 1989 | - Simple to set up and operate<br>- Rates and temporal variation in acid generation and sulphide oxidation can be determined.<br>- Effects of bacterial activity can be assessed<br>- Can be used to assess AMD control options such as blending and submerged deposition<br>- Large scale tests can assess larger waste rock particles and can evaluate internal weathering and chemical changes waste rock and AMD potential.<br>- Accepted method in both Canada and the U.S.A. | - Time period to complete test can be lengthy and hence, costly relative to humidity cells.<br>- Interpretation of results can be complex and the accuracy of the results depends on how well test conditions reflect field conditions.<br>- May experience problems with uneven solution application and channelling.<br>- Not practical for a large number of samples. |
| ***Soxhlet Extraction***<br><br>**Renton et al. 1988**<br>**Sobek et al. 1978** | - Simple to operate<br>- Rapid test procedure<br>- Options may be tested<br>- Easy to interpret (model) results | - geochemistry may be altered<br>- oxidations tend to be aggressive<br>- unnatural conditions<br>- bacterial, temperature, pH effects cannot be determined |

## Table III: 1.2-2 (Continued)

## CHARACTERISTICS OF THE DYNAMIC AMD TESTS

| Test Type | Advantages | Disadvantages |
|---|---|---|
| *Stirred Reactor Studies*<br><br>B.C. AMD Task Force 1989<br>Duncan and Bruynesteyn 1979<br>Filipek et al. 1991<br>Halbert et al. 1983<br>Lawrence et al. 1989<br>Scharer et al. 1991<br>Scharer & Nicholson 1991 | - Amenable to fundamental studies (surficial reaction rate)<br>- Environmental factors are easily assessed:<br>i) oxygen concentration<br>ii) temperature<br>iii) pH<br>iv) specific surface area<br>v) bacterial activity<br>- Allows multilevel factorial statistical design<br>- Control actions (submersion, oxygen exclusion, bacterial inhibition) may be evaluated | - tend to overestimate reaction rates (ideal rate)<br>- cannot be used to evaluate effect of moisture content on oxidation<br>- may be oxygen limited<br>- secondary mineralization may be affected<br>- complex data interpretation and modelling |
| *Stationary Bed Test Studies*<br>Reactions in stationary solid columns<br><br>Bradham and Caruccio 1991<br>B.C. AMD Task Force 1989<br>Caruccio 1968<br>Caruccio et al. 1981<br>Hood and Oerter 1984<br>Ritcey 1989<br>Ritcey and Silver 1982<br>Sobek et al. 1978 | - simulates natural conditions (including submerged conditions)<br>- simple to operate<br>- environmental factors can be assessed<br>- gives overall acid generation per unit mass of waste rock<br>- easy to monitor<br>widely used in US and Canada<br>- control actions may be evaluated | - confounds kinetics with transport phenomena<br>- may be diffusion limited<br>- bacterial acclimatization may be difficult<br>- surfaces are undefined<br>- complex data interpretation and modelling<br>- *may not represent field conditions* |

## TABLE III: 2.2-1

General classifications'of monitoring options for waste rock piles (ChIII:2.2), directions to other sections of the main document with related information and the appropriate Technical Summary Notes

| Monitoring category | Pre-Operation | Operating | Closure | Other Sections | Technical Summary Notes |
|---|---|---|---|---|---|
| **Rock** Characterization | | | | | |
| Chemical characterization | | | | | |
| Elemental Content | P | I | I$^2$ | ChIII:1.1:2 | 6.1 |
| Mineralogy/Mineral Forms | P | I | I$^2$ | ChIII:1.1.2 | 6.2 |
| AMD Assesment Tests | | | | | |
| Static Tests | P | I | I$^2$ | ChIII:1.2.1, 1.3 | A - -<br>- I - - 2 . 1 |
| Dynamic Tests | P | I | I$^2$ | ChIII: 1.2.2, 1.3 | 2.2 |
| Physical Stability | | | | | |
| Hardness | SD | I | I$^2$ | ChIII:1.1.2 | 11.1 |
| Weathering | SD | I | I$^2$ | ChIII:1.1.2 | 11.2 |
| Particle Size Distribution | SD | I | I$^2$ | ••• | 6.3 |
| **Water** | | | | | |
| Water Quality | P | P | P | ChIII:2.5, ChIV:2.0 | 13.0 |
| Monitoring **Locations** | | | | | |
| Surface flow on Pile | N/A | I | I | ••• | 14.1 |
| Pore Water in Pile | N/A | I | I | ••• | 14.2.1, 14.2.2 |
| Seepage to Surface of Pile | N/A | I | I | ••• | 14.1 |
| Groundwater | P | SD/I | I | ChIV:2.2 | 8.0, 14.3 |
| Toe Discharge / Collection Ditch | N/A | P | P | ChII: 1.4, ChIII:2.5, ChIV:2.1 | 14.1 |
| Inflow to Treatment Facility | N/A | P | P | ChIII:2.5, ChIV:2.1 | 14.1 |
| Natural Receiving Waters | P | P | P | ChIII:2.5, ChIV:2.1 | 14.1 |
| Flow Monitorine (Each Station) | P | P | P | ChIV:1.0 | 7.0 |
| **Other** Categories | | | | | |
| Structural Stability | P | P | P | ChIII:2.6 | 12.0 |
| Gas Comoosition | N/A | I | I | --- | 3.1, 3.2 |
| Internal Temperature Profile | N/A | I | I | --- | 3.6 |
| Permeability | | | | | |
| Water Permeability | N/A | I | I | ••• | 3.4.1 |
| Air permeability | N/A | I | I | ••• | 3.4.2 |
| Oxygen Diffusion (Covers) | N/A | N/A | SD/I | --- | 3.2 |
| Porosity | N/A | I | I | --- | 3.5 |
| Water Content | N/A | | I | --- | 14.2.1 |
| Infiltration | N/A | I | I | ••• | 3.3 |
| Meteorology | SD | SD | SD | ••• | 10.0 |
| Geophysical Techniques | N/A | SD/I | SD/I | ••• | 8.0 |
| Bacterial Monitoring | N/A | I | I | --- | 3.7 |
| Collection Options | | | | --- | 5.1, 14.3.2 |

Notes

[1] **Classifications** will **vary** in priority on a site-specific basis. Multiple classifications are in order of most **common** priority.

[2] Additional predictive AMD assessment should be completed prior to decommissioning. These procedures would then be considered investigative component3 for long-term monitoring.

# TABLE III: 2.3-1

General classifications[1] of monitoring options for tailings (ChIII:2.3), directions to other sections of the main document with related information and the appropriate Technical Summary Notes.

| Monitoring Category | Pre-Operation (Pilot Plant) | Operating | Closure | Other Sections | Technical Summary Notes |
|---|---|---|---|---|---|
| **'ailings Solids** | | | | | |
| Chemical characterization | | | | | |
| Elemental Content | P | SD | $I^2$ | ChIII:1.1.2 | 6.1 |
| Mineralogy/Mineral Forms | P | I | $I^2$ | ChIII:1.1.2 | 6.2 |
| AMD Assesment Tests | | | | | |
| Static Tests | P | I | $I^2$ | ChIII:1.2.1, 1.3 | 2.1 |
| Dynamic Tests | P | I | $I^2$ | ChIII:1.2.2, 1.3 | 2.2 |
| Particle Size Distribution | P | I | $I^2$ | --- | 6.3 |
| Compaction | N/A | P/SD | P/SD | --- | 8.5 |
| **Water Quality** | | | | | |
| Water Quality | N/A | P | P | ChIII:2.5, ChIV:2.0 | 13.0 |
| Water Monitoring Locations | | | | | |
| Pore Water in TMF | N/A | SD/SI | SD/SI | ChIV:3.6.1 | 14.2.3, 14.2.4, 14. |
| Embankment Seepage | N/A | P | P | --- | 14.1 |
| Groundwater | N/A | SD/P | P/SD | ChIV:2.2 | 8.0, 14.3 |
| Surface Water in TMF$^3$ | N/A | P | P | ChIV:2.1 | 14.1 |
| Side and Bottom Drains | N/A | SD | SD | --- | 14.1 |
| Release to Surface Water | N/A | P | P | ChIII:2.5, ChIV:2.1 | 14.1 |
| Inflow to Treatment | N/A | P | P | ChIII:2.5, ChIV:2.1 | 14.1 |
| Flow Monitoring (Each Station) | N/A | P | P | ChIV:1.0 | 7.0 |
| Colonization of Flooded Tailings | N/A | N/A | SD/P | ChIV:4.2 | 4.2 |
| Plankton Colonization of Flooded TMF | N/A | N/A | SD/P | ChIV:4.1 | --- |
| **Other Categories** | | | | | |
| Structural Stability | N/A | P | P | ChIII:2.6 | 12.0 |
| Gas Composition | N/A | I | SI | --- | 3.1, 3.2 |
| Internal Temperature Profile | N/A | I | SI | --- | 3.6 |
| Permeability | | | | | |
| Water Permeability | N/A | I | SI | --- | 3.4.1 |
| Air Permeability | N/A | I | SI | --- | 3.4.2 |
| Oxygen Diffusion (Covers) | N/A | I | SI | --- | 3.2 |
| Porosity (of deposited tailings) | N/A | I | SI | --- | 3.5 |
| Infiltration | N/A | SD | SD | --- | 3.3 |
| Meteorology | SD | P | SD | --- | 10.0 |
| Geophysical Techniques | N/A | SD/I | SD/I | --- | 8.0 |
| Collection Options | --- | --- | --- | ChIV:3.4.2.7 | 5.2 |

[1] Classifications will vary in priority on a site-specific basis. Multiple classifications are in order of most common priority.

[2] Additional predictive AMD assessment should be completed prior to decommissioning. The-se procedures would then he considered investigative components for long-term monitoring.

[3] Surface water in a TMF concerns ponding in sub-aerial deposition and large volumes in sub-aqueous deposition or flooded TMFs.

## TABLE III: 2.4-1

General classifications[1] of monitoring options for mineworks (ChIII:2.4), directions to other sections of the main document with related information and the appropriate Technical Summary Notes.

| Monitoring Category | Pre-Operation | Operating | Closure | Other Sections | Technical Summary Notes |
|---|---|---|---|---|---|
| **Mine Rock** | | | | | |
| Chemical characterization | | | | | |
| Elemental Content | P | I | I[2] | ChIII:1.1.2 | 6.1 |
| Mineralogy/Mineral Forms | P | I | I[2] | ChIII:1.1.2 | 6.2 |
| AMD Assesment Tests | | | | | |
| Static Tests | P | I/SD | I[2] | ChIII:1.2.1, 1.3 | 2.1 |
| Dynamic Tests | P | I/SD | I[2] | ChIII:1.2.2, 1.3 | 2.2 |
| Physical Stability | | | | | |
| Hardness | SD | I | I[2] | ChIII:1.1.2 | 11.1 |
| Weathering | SD | I | I[2] | ChIII:1.1.2 | 11.2 |
| **Water** | | | | | |
| Water Quality | P | P | P | ChIII:2.5, ChIV:2.0 | 13.0 |
| Water Monitoring Locations | | | | | |
| Outflows to Surface[3] | P | P | P | ChIII:2.5, ChIV:2.1 | 14.1 |
| Flooded Mineworks | N/A | N/A | P/SD | ChIV:2.1 | |
| Groundwater | P | SD/P | P/SD | ChIV:2.2 | 8.0, 14.3 |
| Pit or underground sumps | N/A | P/SD | N/A | --- | 14.1 |
| Inflow to Treatment Facilities | N/A | P/SD | P/SD | ChIII:2.5, ChIV:2.1 | 14.1 |
| Flow Monitoring (Each Station) | P | P | P | ChIV:1.0 | 7.0 |
| **Other Categories** | | | | | |
| Structural Stability | P | P | P | ChIII:2.6 | 12.0 |
| Water Permeability | | I | I | --- | 3.4.1 |
| Meteorology | SD | P | P | --- | 10.0 |
| Geophysical Techniques | I | I/SD | SD/I | --- | 8.0 |
| Collection Options | --- | --- | --- | --- | 5.1 |

Notes

[1] Classifications will vary in priority on a site-specific basis. Multiple classifications are in order of most common priority.

[2] Additional predictive AMD assessment should he completed prior to decommissioning. These procedures would then be considered investigative components for long-term monitoring.

[3] This includes drainage from exploration trenches and adits, etc.

# TABLE III: 2.5-1

Parameters which can be determined using continuous time probes and data loggers.

| Data | Unit | Range[1] | Accuracy[1] |
|---|---|---|---|
| Temperature | °C | -20 to 45 | ± 0.2 |
| Dissolved Oxygen | % Saturation | 0 to 200 | ± 2 |
| | mg/L | 0 to 20 | ± 0.2 |
| pH Value | pH units | oto 14 | ± 0.2 |
| Conductivity | µs/cm | 0 to 100,000 | ± 0.5% of reading |
| Turbidity* | NTU | 0 to 1,000 | ± 5% of reading |
| Salinity | ppt | 0 to 70 | ± 1% of reading |
| ORP | mV | -999 to 999 | ± 20 |
| Ammonia | mg/L - N | 0 to 200 | ± 10% of reading |
| Ammonium-Nitrogen | mg/L - N | 0 to 20 | ± 0.2 |
| Nitrate Nitrogen | mg/L - N | 0 to 200 | ± 10% of reading |
| Hydrocarbons | ppm | 0 to 20,000 | ± 10% of reading |

[1] Range and accuracy as reported are typical of these instruments. However both range and accuracy vary with the instrument manufacturer.

[2] In terms of adsorption of ultra-violet light.

# TABLE IV: 1.1-1

## Comparison of common methods for determining stream stage.

| **Method** | Advantages | **Disadvantages** | **instrument Cost[1]** |
|---|---|---|---|
| Float/Chart Recorder | • Very accurate stage measurement<br>• Records stage on paper which may be preferred in some cases | • Open water season only<br>• Requires installation of a stilling well<br>• Requires manual conversion of chart data to digital data<br>• Requires periodic (e.g., monthly) replacement of chart | $2,000 - 6,000 |
| Pressure Transducer and Data Logger | • Very accurate; typical accuracy is 0.10 % of the total measurement range.<br>• Works under ice cover[2] | • Must be located near bottom of stream so installation in deep or contaminated water may be problematic | $4,000 - 8,000 |
| Ultrasonic Sensor and Data Logger | • Mounting over the stream may be an advantage in very deep channels<br>• Not mounted in the channel so instrument and worker exposure to potentially hazardous environment is avoided<br>• Very accurate; typical accuracy is 0.30 % of the total measurement range. | • Mounting over the stream may not be practical<br>• Does not work where surface of the stream is highly turbulent or ice-covered | $4,000 - 8,000 |
| Bubbler and Data Logger | • Highly suited for corrosive environments because the only contact with flow is the bubbler tube<br>• Very accurate; typical accuracy is 0.50 % of total measurement range<br>• Works under ice cover[2] | • **Battery** life may be slightly less than other sensor/data logger methods<br>• Must be located near bottom of stream so installation in deep or contaminated water may be problematic | $4,000 - 8,000 |

[1] **Cost** is for the instrument only. Additional costs include installation and the establishment of rating curves.

[2] These instruments may be used under ice but separte rating curves should be developed for ice conditions.

# TABLE IV: 2.1-1

Detection limits for trace elements’ in water for various methods.

| Element | FAAS | ICAP/AES | ‘GFAAS- | HVAAS | CVAAS |
|---|---|---|---|---|---|
| Aluminum | 0.10 | 0.20 | 0.005 | | |
| Arsenic | 5.0 | 0.20 | | 0.0001 | |
| Cadmium | 0.005 | 0.010 | 0.0002 | | |
| Chromium | 0.01 | 0.015 | 0.001 | | |
| Copper | 0.005 | 0.010 | 0.001 | | |
| Iron | 0.03 | 0.030 | 0.005 | | |
| Lead | 0.05 | 0.050 | 0.001 | | |
| Mercury | | | | | 0.00005” |
| Nickel | 0.01 | 0.020 | 0.001 | | |
| Selenium | | 0.20 | | 0.0005 | |
| Zinc | 0.005 | 0.005 | 0.001 | | |

All numbers are expressed as milligrams per litre (mg/L).

‘Note: Arsenic and selenium are non-metals.

‘Lower detection limits are available by special request (0.000005 mg/L).

CVAAS • Cold-Vapour Atomic Absorption Spectrophotometry.

FAAS • Flame Atomic Absorption Spectrophotometry.

GFAAS • Graphite-Furnace Atomic Absorption Spectrophotometry.

HVAAS • Hydride-Vapour Atomic Absorption Spectrophotometry.

ICAP/AES • Inductively-Coupled Argon Plasma/Atomic Emission Spectrophotometry.

## TABLE IV: 2.1-2

Summary of recommended procedures for sample collection, preservation, and storage’.

| Determination | Collection Method | Minimum Sample Size | Container | Preservation/Storage | Maximum Storage Time’ |
|---|---|---|---|---|---|
| Alkalinity | Grab/Pump | 200 mL | P, G | Cool 2-5°C no air space | 24 h |
| BOD | Grab/Pump | 1 L | P, G | Refrigerate, keep in dark | 6 h |
| Carbon, Organic, total | Grab/Pump | 100 mL | P, G | Analyze immediately or refrigerate and add HCL to pH <2 | 7 d |
| Chlorophyll | Grab/Pump | 500 mL | P, G | Refrigerate, keep in dark | 30 d |
| Conductivity | Grab/Pump | 500 mL | P, G | Refrigerate | 28 d |
|  | Field Meter/Continuous Probe | ---- | ---- | None: in situ measurement | ---- |
| Hardness | Grab/Pump | 100 mL | P, G | Add $HNO_3$ to pH <2 | 6 months |
| Metals: general | Grab/Pump | 500 mL | P(A), G(A) | Filter dissolved metals immediately, add $HNO_3$ to pH <2 | 6 months |
| mercury | Grab/Pump | 500 mL | P(A), G(A) | Add $HNO_3$ to pH <2, refrigerate | 28 d |
| Nitrogen: ammonia | Grab/Pump | 500 mL | P, G | Analyze immediately or add $H_2SO_4$ to pH <2 | 28 d |
|  | Field Meter/Continuous Probe | ---- | ---- | None: in situ measurement | ---- |
| nitrate | Grab/Pump | 100 mL | P, G | Analyze as soon as possible or refrigerate | 7 d |
|  | Field Meter/Continuous Probe | ---- | ---- | None: in situ measurement | ---- |
| nitrate + nitrite | Grab/Pump | 200 mL | P, G | Add $H_2SO_4$ to pH <2, refrigerate | 28 d |
| nitrite | Grab/Pump | 100 mL | P, G | Analyze as soon as possible or refrigerate | 48 h |
| organic, Kjeldahl | Grab/Pump | 500 mL | P, G | Add $H_2SO_4$ to pH <2, refrigerate | 7 d |
| Oxygen | Grab/Pump | 300 mL | G, BOD bottle | Titration may be delayed after acidification | 8 h |
|  | Field Meter/Continuous Probe | ---- | ---- | None: in situ measurement | ---- |
| pH | Grab/Pump | 100 mL | P, G (B) | Analyze immediately | 2 h |
|  | Field Meter/Continuous Probe | ---- | ---- | None: in situ measurement | ---- |
| Phosphate | Grab/Pump | G(A) | 100 mL | For dissolved phosphate filter immediately; refrigerate | 48 h |

# TABLE IV: 2.1-2

Summary of recommended procedures for sample collection, preservation, and storage[1].

| Determination | Collection Method | Minimum Sample Size | Container | Preservation/Storage | Maximum Storage Time |
|---|---|---|---|---|---|
| Salinity | Grab/Pump | G, wax seal | 240 mL | Analyze immediately or use wax seal | 6 months |
| Sulphate | Grab/Pump | P, G | 500 mL | Refrigerate | 28 d |
| Sulphide | Grab/Pump | P,G | 100 mL | Refrigerate: add 4 drops 2 N zinc acetate/100 mL; add NaOH to pH >9 | 28 d |
| Suspended solids | Grab/Pump | P,G | 500 mL | Refrigerate | 7 d |
| Temperature | Grab/Pump | P,G, | 100 mL | Analyze immediately | N.S. |
|  | Field Meter/Continuous Probe | ---- | ---- | None: in situ measurement | ---- |
| Turbidity | Grab/Pump | P, G |  | Analyze same day; or refrigerate in dark | 24 h |

---

[1] Compiled from a number of sources: Environment Canada 1993, U.S. EPA **1987a,** 1992b.

Storage times are conservative. Consult these references or an acredited laboratory for further information.

Refrigerate = storage at 4” C, in the dark. P = polyethylene plastic; **G** = glass G(A) or P(A) = rinsed with 10% $HNO_3$; G(B) = borosilicate glass, G(S) or P(S) = rinsed with organic solvents and used with tellon-lined lid. **N.S.** = no storage allowed; analyze immediately.

## TABLE IV: 2.1-3

Specific information requirements in field notes for selected parts of a monitoring program (Adapted from BC AMD 1992).

| Monitoring Activity | Information Requirements |
|---|---|
| Surface water | Sampling date and time (which may allow water quality to be related to stream flow data or to known upstream events) |
| | Air and water temperature |
| | Weather conditions (to evaluate the effect of storm events) |
| | Location of discharges, seeps, or ephemeral streams; observations about bank slumpages or other potential sources of environmental contamination |
| Groundwater | Number of well volumes removed during purging |
| | General description of the appearance of the water (color, suspended sediment, precipitates formed upon contact with the atmosphere, odor) |
| | Field water quality measurements (conductivity, pH, Eh, temperature) |
| Sediments | Basic details on the equipment used during sampling |
| | Sediment type and texture |
| | Sediment color |
| | Presence, type and strength of odors |
| | Depth of penetration of the core or grab sampler |
| | Degree of leakage or surface disturbance |
| | Presence of large debris or benthic fauna |
| | For streams include characteristics such as average channel width, wetted width, gradient channel features, bank type, bank stability, bank height, confinement, stage, flood signs and estimate discharge rate (see TSN #) |
| Fish | For stream and river sampling, habitat notes should be recorded (see Habitat Inventory Fact Sheet) |
| | For lakes, netting times, weather and water conditions (including results of field measurements such as pH, dissolved oxygen, Secchi disc depth), mesh type and size, net size, net depth and locations |
| | Data from field measurements (weight, fork length, condition) and identification (by individual fish) for any sample retaining for laboratory analysis (scales, stomach contents) |
| Benthic invertebrates and periphyton | Number and location of replicated samples |
| | Type of sampler |
| | Sampling intensity |
| | Microhabitat information (substrate type, velocity of flow, and depth of water) if stratified design is used |

# TABLE IV: 2.2-1

Drilling methods for monitoring wells.

| Type | Advantages | Disadvantages |
|---|---|---|
| Hollow Stem Auger | • No drilling fluid is used, eliminating contamination by drilling fluid additives<br>• Formation waters can be sampled during drilling by using a screened auger or advancing a well point ahead of the augers<br>• Formation samples taken by split-spoon or core-barrel methods are highly accurate<br>• Natural gamma-ray logging can be done inside the augers<br>• Hole caving can be overcome by setting the screen and casing before the augers are removed<br>• Fast<br>• Rigs are highly mobile and can reach most drilling sites<br>• Usually less expensive than rotary or cable tool drilling | • Can be used only in unconsolidated materials<br>• Limited to depths of 100 to 150 ft (30.5 to 45.7 m)<br>• Possible problems in controlling heaving sands<br>• May not be able to run a complete suite of geophysical logs |
| Direct Rotary | • Can be used in both unconsolidated and consolidated formations<br>• Capable of drilling to any depth<br>• Core samples can be collected<br>• A complete suite of geophysical logs can be obtained in the open hole<br>• Casing is not required during drilling<br>• Many options for well construction<br>• Fast<br>• Smaller rigs can reach most drilling sites<br>• Relatively inexpensive | • Drilling fluid is required and contaminants are circulated with the fluid<br>• Drilling fluid mixes with the formation water and invades the formation and is sometimes difficult to remove<br>• Bentonitic fluids may absorb metal and may interfere with other parameters<br>• During drilling, no information can be obtained on the location of the water table and only limited information on water-producing zones<br>• Formation samples may not be accurate |

# TABLE IV: 2.2-l

## Drilling methods for monitoring wells.

| Type | Advantages | Disadvantages |
|---|---|---|
| Air Rotary | . No water-based drilling fluid is used, eliminating contamination by additives<br>. Field analysis of water blown from the hole can provide information regarding changes for some basic water-quality parameters such as chlorides<br>. Capable of drilling to any depth<br>. Formation sampling is excellent in hard, dry, formations<br>. Formation water blown out of the hole makes it possible to determine when the first water-bearing zone is encountered<br>. Can be used in both unconsolidated and consolidated formations<br>. Fast | . Casing is required to keep the hole open when drilling in soft, caving formations below the water table<br>. When more than one water-bearing zone is encountered and hydrostatic pressures are different, flow between zones occurs during the time drilling is being completed and before the borehole can be cased and grouted properly<br>. Relatively more expensive then other methods<br>. May not be economical for small jobs |
| Cable Tool | . Only small amounts of drilling fluid are required (generally water with no additives)<br>. Can be used in both unconsolidated and consolidated formations; well suited for extremely permeable formations<br>. Can drill to depths required for most monitoring wells<br>. Highly representative formation samples can be obtained by an experienced driller<br>. Changes in water level can be observed<br>. Relative permeabilities for different zone can be determined by skilled drillers<br>. A good seal between casing and formation is virtually assured if flush-jointed casing is used<br>. Rigs can reach most drilling sitesand are relatively inexpensive | . Minimum casing size is 4 in (102 mm)<br>. Steel casing must be used<br>. Cannot run a complete suite of geophysical logs<br>. Usually a screen must be set before a water sample can be taken<br>. Slow |

## TABLE IV: 2.2-1

Drilling methods for monitoring wells.

| Type | Advantages | Disadvantages |
|---|---|---|
| Becker Hammer | • Fast, even in coarse-grained, cohesionless soils<br>• Reverse air circulation prevents mixing of cuttings | • Can only drill locations with good road access<br>• Cannot penetrate large boulders without coring or blasting |
| Diamond Drill | • Capable of penetrating very had bedrock<br>• Capable of penetrating boulders in overburden<br>• Continuous core recovery possible in competent ground<br>• Small rigs can be helicopter portable | • Requires drill mud for efficient advance<br>• Relatively slow in overburden |

## TABLE IV: 2.2-2

Generalized ground-water sampling protocol (from US EPA 1991).

| Step | Goal | Recommendations |
|---|---|---|
| Hydrological Measurements | Establishment of nonpumping water level. | Measure the water level to ±0.3 cm (±0.01 ft). |
| Well Purging | Removal or isolation of stagnant $H_2O$ which would otherwise bias representative sample. | Pump water until well purging parameters (e.g., pH, T, $\Omega^{-1}$, Eh) stabilize to ± 10% over at least two successive well volumes pumped. |
| Sample Collection | Collect samples at land surface or in well-bore with minimal disturbance of sample chemistry. | Pumping rates should be limited to approximately 100 mL/min for volatile organics and gas-sensitive parameters. |
| Filtration/Preservation | Filtration permits determination of soluble constituents and is a form of preservation. It should be done in the field as soon as possible after collection. | Filter: Trace metals, inorganic anions/cations, alkalinity. Do not filter: TOC, TOX, volatile organic compound samples. Filter other organic compound samples when required. |
| Field Determinations | Field analyses of samples will effectively avoid bias in determinations of parameters/constituents which do not store well: e.g., gases, alkalinity, pH. | Samples for determinations of gases alkalinity and pH should be analyzed in th field if at all possible.. |
| Field Blanks/Standards | These blanks and standards will permit the correction of analytical results for changes which may occur after sample collection: preservation, storage, and transport. | At least one blank and one standard for each sensitive parameter should be made u in the field on each day of sampling. Spiked samples are also recommended for good QA/QC. |
| Sampling Storage/Transport | Refrigeration and protection of samples should minimize the chemical alteration of samples prior to analysis. | Observe maximum sample holding or storage periods recommended by the Agency. Documentation of actual holding periods should be carefully performed. |

## TABLE IV: 2.2-3

Comparison of methods for determining water surface in monitoring wells.

| Method | Advantages | Wadvantages | Accuracy | Equipment Cost |
|---|---|---|---|---|
| teel Tape | • Simple and reliable method<br>• Inexpensive | • Impractical for wells > 30 m deep<br>• Must be removed from well completely for multiple readings<br>• Multiple readings (typically 3) are required to ensure accuracy | ± 3 mm | $200-600 |
| lectric Drop Line | • May be combined with other data collection including temperature and conductivity<br>• May be used in wells to 500 m deep<br>• Quicker for multiple readings than steel tape method | • Cable has a tendency to become kinked over time<br>• Least accurate in shallow wells | ± 25 mm | $500 - 1,500 |
| ‘ressure Transducer nd Data Logger | • May be combined with other data collection including temperature, oxygen, **pH,** and conductivy<br>• Can collect data in remote areas unattended<br>• Provides continuous monitoring at user-defined time intervals | • Download of data usually requires taking a portable computer in the field<br>• Relatively high cost on a per well basis | ± 1-15 mm | $4,000 - 10,000 (per well) |

# TABLE IV: 2.2-4

## Monitoring well sampling equipment.

| Type : | Approximate Cost | Advantages : | Disadvantages |
|---|---|---|---|
| Bailers | $50 to $500 | *Inexpensive<br>*Simple to operate<br>*Highly portable<br>*Can lift from any depth | •Low sample quality<br>•Can be time consuming for large purge volumes |
| Inertial Pumps | $20 to $50 | *Inexpensive<br>*Simple to operate<br>*Fast where water levels are close to surface | •Low portability requires dedication to a single installation<br>•Low pumping rates in deep water level situations |
| Bladder Pumps | $1,000 to $2,500 | *High quality water samples<br>*Moderately fast | *Expensive, especially for dedicated systems<br>•Low portability, even for dedicated systems |
| Submersible Pumps | $1,000 to $5,000 | *High pumping rates<br>*Battery powered models can fit in 50 mm wells | •Create considerable sample agitation<br>•High lift, high volume pumps require large diamete wells and large power sourc<br>•Metal construction may interfere with sample water quality |
| Suction Lift Pumps (direct line, centrifugal, peristaltic pumps) | $100 to $500 | *Simple<br>*Direct line and centrifugal pumps can achieve high pumping rates | *Peristaltic pumps slow<br>*Limited lift<br>*Suction can cause degassin and changes in water chemistry<br>*Some sample mixing and oxidation |
| Air Lift | $100 to $500 | *Fast<br>•Simple<br>Can be used for well development | *Aeration of sample causes changes in chemistry<br>•In general not appropriate for water quality sampling |

## TABLE IV: 2.2-4

Monitoring well sampling equipment.

| Type | Approximate Cost | Advantages | Disadvantages |
|---|---|---|---|
| Gas Displacement Pumps | $500 to $2,000 | •Use air lift principle with refinements to limit mixing and aeration<br>•Can be constructed of Teflon sample-contact parts<br>*Multiple functions | •Relatively expensive<br>*Require compressed gas source<br>•Sophisticated construction |
| Gas Piston Pumps | $500 to $2000 | •Can sample up to 150 m | •Generally low pumping rate<br>*Metal parts may affect wate quality |
| Packer Pumps | $1,500 to $3,000 | •Isolate portions of the standpipe, reducing purge volumes<br>*Can be used in combination with a number of different samplers | •Add expense, complexity, and material to the sampling equipment |

## TABLE IV: 3.4-1

Information requirements for the Environmental Characterization or Preliminary Survey Phase of a sediment monitoring program.

**Hydrologic (Chapter IV: 1.0) and Water Quality (Chapter IV: 2.0) Information**

- quality and quantity of surface inflows and outflows to the system
- quality and quantity of groundwater influences to the system
- bathymetry (water depth distribution)
- velocity and direction of currents
- plume delineation modelliag or dye study

**Preliminary Sediment** Characterization

- complete a literature review for historical sediment data for the specific study area and/or the region.
- identify zones of erosion, transport and deposition
- characterize the sedimentation rates (literature estimaates or sedimentation trays)
- identify sediment types (fines, gravel, etc.), distribution and thickness

**Anthropogenic Considerations**

- identify and locate project source influences (both AMD and non-AMD) sediment quality and quantity
    - locate waste disposal or storage sites and determine potential sediment contaminant pathways
    - locate sewage treatment facilities and determine potential sediment contaminant pathways
    - identify sites of surface disturbance and the potential for influencing sedimentation rates
    - identify high risk areas for spill contamination (e.g., dyke wall failure, etc.)
- identify past and present additional potential influences to sediment quality and quantity
    - e.g., forestry, road construction, mining activities, industrial activities, urban centres
    - specifically identify ARD potential for these possible influences
    - compile a potential sediment contaminant list for these possible influences
    - locate areas of historical spills or disposal

**Geochemical Relationships**

- characterize surficial geology and soils in relation to pre-operational sediment geochemistry

**Climatic Factors**

- wind conditions (direct effect or indirectly through seiches)
- seasonal influences on temperature, precipitation, solar radiation, etc.

# TABLE IV: 3.4-2

A) Sediment particle sizes class names (modified Wentworth-Udden Scale) and B) analytical techniques for determining particle size (From Horowitz 1991).

**a)**

| Class Name | Millimetres | Micrometres |
|---|---|---|
| Boulders | > 256 | - |
| Large Cobbles | 256 - 128 | - |
| Small Cobbles | 128 - 64 | - |
| V. Coarse Gravel | 64 - 32 | - |
| Coarse Gravel | 32 - 16 | - |
| Medium Gravel | 16 - 8 | - |
| Fine Gravel | 8 - 4 | - |
| V. Fine Gravel | 4 - 2 | - |
| V. Coarse Sand | - | 2000 - 1000 |
| Coarse Sand | - | 1000 - 500 |
| Medium Sand | - | 500 - 250 |
| Fine Sand | - | 250 - 125 |
| V. Fine Sand | - | 125 - 62.5 |
| Coarse Silt | - | 62.5 - 31 |
| Medium Silt | - | 31 - 16 |
| Fine Silt | - | 16 - 8 |
| V. Fine Silt | - | 8 - 4 |
| Coarse Clay | - | 4 - 2[a] |
| Medium Clay | - | 2 - 1 |
| Fine Clay | - | 1 - 0.5 |
| V. Fine Clay | - | 0.5 - 0.25 |

**b)**

| Analysis Techniques |
|---|
| > 63 µm Fraction<br>Sieving<br>Visual Accumulation |
| < 63 pm Fraction<br>Physical Separations<br>Pipette Analysis<br>Decantation<br>Bottom Withdrawal<br>Centrifugation<br>Air Elutriation<br>Measurements<br>Electrozone Counters<br>Sedigraph<br>Microscopy<br>Hydrometers<br>Image Analyzers |

[a] Many sedimentologists consider the silt/clay break to occur at 2 µm rather than at 4 µm.

# TABLE IV: 3.4-3

Summary of recommended dredge/grab samplers for collection of sediment samples.
See TSN 4.5 for additional information.

| Sampler | Surface Area/ Sample Depth | Sediment Type | Advantages | Limitations | Recommended For: |
|---|---|---|---|---|---|
| **Birge-** Ekman Dredge | 0.023 or 0.052 m$^2$<br>0.0 - 30 cm | Soft sediments such as silt and muds. May be modified (weights added) to sample fine sands. Requires minimal to no current. | Easy to operate by hand without a winch and easily transported (easily backpacked). Handle attachment for shallow operation and hinged flaps at top reduce washout, shock waves and substrate disturbance. | Will not penetrate hard substrates. Jaws easily blocked by light debris preventing proper closure. Inefficient in deep water or moderate current. | Soft sediments from shallow to moderate water depths. When sampling from the ice surface, small boats, or wading. Use restricted to sample areas with low water currents. |
| Petite Ponar Grab | 0.023 m$^2$<br>0.0- 10 cm | Moderately hard sediments such as sand, silt and mud. Will not penetrate clay and less efficient in soft sediments and gravel. | Good penetration for a small grab. Side plates and screens reduce washout, shock waves and substrate disturbance, Can he operated by without a winch. | Less easily field packed than Ekman. Jaws can be blocked heavy debris (stones etc.). Not efficient in Waler velocities over one meter per second velocity. | Firm to hard sediments sampled from the ice surface or small boats with or without a hoist system. May sample in areas of moderate current. |
| Standard **Ponar** Grab | 0.052 m$^2$<br>0.0 - 10 cm | Coarse and firm sediments such as clay, hard pan, sand, gravel. Samples fines less efficiently. | Better penetration and jaws more resistant to blockage than above described grabs. Side plates and screens reduce washout, and substrate disturbance. | Heavy sampler requiring a solid platform (larger boat or ice) and a winch or hoist. Jaws may be blocked by heavy debris (stone etc.). | Hard sediments, collected from the ice surface or a boat using a pulley or winch system. May sample in areas of moderate current. |
| Shipek Grab Sampler | 0.04<br>0.0 - 10 cm | Soft fine grained to sandy sediments. | Can collect relatively undisturbed samples from sloping bottoms at any depth. Easy access to sample for subsampling or observations when sampling bucket removed. Less susceptible to washout compared to the other grabs. | Poor for packed sediments. Triggering mechanism sensitive requiring cautious handling. | Fine sediments collected from the ice surface or a boat using a pulley or winch system. May sample in areas of moderate current. **Ideal** for sampling sloping bottoms. |
| Van Veen Grab | 0.06 - > 0.1 m$^2$ 0.0<br>30 cm | Soft fine sediments to sands. | Jaws close tightly; samples most sediment types; comes in a range of sizes. Collects a large volume of sample. | A very heavy grab that requires a solid sampling platform (ice or large boat) and a power winch. Jaws may be blocked by heavy debris. | Excellent for sampling in deep water where strong underwater currents are present and when a solid sampling platform and winch are available. |

# TABLE IV: 3.4-4

Summary[1] of the most common commercially available core samplers for collection of sediment samples.
See TSN 4.5 for additional information.

| Sampler | Interior Diameter | Habitat | Advantages | Limitations | **Recommended** for: |
|---|---|---|---|---|---|
| KB Corer Gravity Corer | 5.08 cm core | Rivers, lakes, estuaries; good for soft sediments only, especially silt and clay. | Standard and heavy models available based on a wide variety of modifications. Standard version (head wt. = 9 kg) may be used without a winch. Minimum shock wave compared to other gravity corers. | Messenger **operated** valve version sometimes fails to trigger. Winch required if additional tins, weights and long core tube used. | Soft sediments in shallow to medium water depths and when a winch is not available. |
| **Balcheck** Single and Multiple Tube Gravity Corer | 5.08 or 7.62 cm core | Rivers, lakes, estuaries; good for soft sediments only, especially silt and clay. | Good penetration of soft sediments; multiple (four) core tube available; wide variety of modifications available. | Heavy (38 kg); requires boat and winch to operate; does not retain sand unless heavy core catchers are used; pressure wave may disturb **water/sediment** interface. Check valves work automatically to prevent loss of sample. | Good for sampling in **soft** sediment; used in shallow or deep waters. |
| Phleger Gravity Corer | 3.5 cm core | Wide range of sediments from soft and sandy to semi-compacted. | Will penetrate peats and root **matts** in shallow lakes and marshes. Exhibits relatively stable vertical penetration and does not require a winch. | The small sample sizes resulting from the narrow core tubes is a distinct disadvantage. | Best utilized for marshes and sediments with root matts where other corers have difficulty penetrating. |
| Box Corer | **surface** area of 100 **cm$^2$ depth** of **20** cm | All sediment types in large waterbodies. | Good penetrations in soft sediments; causes little disturbance to water/sediment interface; may be deployed from ships or other platforms. | Gravity operated; heavy; requires boat and large winch to operate; diver collected cores preferable to ensure that sample is taken vertically (not on an angle). | Good for sampling in soft sediment; used in shallow or deep **waters;** collects qualitative and quantitative samples. |
| **Hand-** Operated Corer | variable, generally shallow corer with small diameter | All shallow water habitats, sampled by hand or by diver. | Small sample volume; can be used with **extension handles** of one to three **meters;** used with pipe fitting for driving from a pontoon boat, dock or bridge. | Restricted **to depth** of extension handle and **difficult** to **operate** over 2 - 3 m deep. Must lower slowly due to restricted flow-through . | Effective for shallow water sampling; **with divers, can be used for deeper** water (up to 100 feet). |

---

[1] Comparisons are between corers and not other sediment sampling devices.

## TABLE IV: 3.4-5

List of the potential parameters of interest in a **routine** sediment bulk contaminant characterization monitoring program.

| Parameter | Analytical Method[2] | Dry Mass[3] | Recommended Minimum Parameter List[1] | | | |
|---|---|---|---|---|---|---|
| | | | **Preliminary Survey** (See Section IV: 3.4.1) | **Intensive Baseline** (See Section IV: 3.4.2.9) | **Operational Monitoring** (See Section IV: 3.4.2.9) | **Decommissioning Monitoring** (See Section IV: 3.4.2.9) |
| Physical logging | Visual | N/A | R | R | R | R |
| Wet weight / Dry weight ratio | Laboratory weighing | N/A | R | A | R | R |
| pH | Field Meter | | | A | A | A |
| Redox Potential (Eh) | Field meter | | | R | A | A |
| Sediment grain size | See Table: 8.4-2 | 10 g to 50 g | R | M, R | M,R | M,R |
| % Loss on Ignition (% LOI) | Combustion @ 550°C | 1 g | R | M,R | M,R | M,R |
| Cation Exchange Capacity (CEC) | ammonium acetate extraction | 10 g | | R | A | A |
| Carbon | | | | | | |
| Total | Leco furnace | 1 g | | R | A | A |
| Organic | acid reacted/ Leco furnace | 1 g | R | M,R | M,R | M,R |
| Inorganic | calculated | --- | | R | A | Sp |
| Nitrogen | | | | | | |
| Ammonia ($NH_4^+$ and $NH_4OH$) | water leach, colourimetric | 1 g[4] | | R | R,Sp | SP |
| Nitrite (NO;) and Nitrate ($NO_3^-$) | water leach, colourimetric | 1 g[4] | | R | A | SP |
| Total Kjeldahl Nitrogen | acid digestion, colourimetric | 1 g[4] | | R | A | sp |
| Phosphorous | | | | | | |
| Total | acid digestion, colourimetric | 1 g | | R | A | SP |
| Inorganic | acid leach, colourimetric | 1 g | | R | A | Sp |
| Organic | calculated | --- | | R | A | s p |
| Sulphur | | | | | | |
| Total Sulphur | Leco furnace | 1 g | | R | A | Sp |
| Sulphate (acid soluble) | $HClO_4$ acid leach, ICP-AES[5] | 1 g | | R | A | Sp |
| Sulphide | distillation, electrode | 1 g | | R | A | s p |

# TABLE IV: 3.4-5

List of the potential parameters of interest in a routine sediment bulk contaminant characterization monitoring program.

| **Parameter** | Analytical Method’ | Dry **Mass**[3] | Recommended Minimum Parameter List’ | | | |
|---|---|---|---|---|---|---|
| | | | **Preliminary Survey** (**See** Seclioo IV: 3.4.1) | **Intensive Baseline** (See **Section** IV: 3.4.2.9) | **Operational Monitoring** (**See Section** IV: 3.4.2.9) | Decommissioning **Monitoring** (See **Section** IV: **3.4.2.9**) |
| Selenium | ICP - AES or Hydride A.A.[6] | 1 g | | R | R,Sp | sp |
| MMLER **Parameters**[7] | | | | | | |
| Arsenic | ICP - AES or Hydride A.A. | 1 g[4] | | M,R | M,R | M,R |
| Copper | ICP - AES | 1 g[4] | | M,R | M,R | M,R |
| Lead | ICP - AES or Graphite Furnace A.A.[8] | 1 g[4] | | M,R | M,R | M,R |
| Nickel | ICP - AES | 1 g[4] | | M,R | M,R | M,R |
| Zinc | ICP - AES | 1 g[4] | | M,R | M,R | M,R |
| Metals | | | | | | |
| - Non-MMLER Metals | ICP - AES | 1 g | | M,R | R | A |
| Mercury | Cold Vapour - A.A.[9] | 0.5 g | | M,R | R, Sp | R, Sp |
| Ore / Process specific parameters | | | | M,R | M,R | M,R |
| e.g. Pb-2 10 | chemical separation, / beta counting | 1 g | | SP | SP | SP |
| PO-210 | alpha spectrometry | 1 g | | SP | SP | SP |
| Ra-226 | chemical separation / alpha counting | 0.5 g | | SP | SP | SP |
| Th-230 | alpha - spectrometry | 0.5 g | | SP | SP | SP |
| Uranium | delayed neutron counting | 2 g | | Sp | Sp | Sp |
| CN: gold cyanide leach facilities | distillation, clourimetric | 1 g | | Sp | Sp | Sp |

[1] Note: the relevent regulatory bodies should be consulted prior to determing the **parameter** list of any routine sediment monitoring program.

M - Minimum requirement: parameters likely to be required by regulators (compliance) in a broad number of jurisdictions.

**R** - Recommended parameter: recommended due to their relevance to **a** broad number of mining operations (diagnostic) or toxicity (e.g.. selenium).
recommended due to relatively inexpensive analytical costs vs their information return (e.g., broad ICP trace element scan).

A - Additional parameters: additional parameters of interest, budget permitting.

Sp - Site specific parameters: parameters classified as compliance or recommended parameters for specific mining sites, ore lypes or milling operations.

[2] The most common analytical method (and usually cheapest) providing moderate to low detection limits is listed first.

[3] Dry mass required to fully utilize detection capabilities of described analytical method.

[4] 1 g sample suitable for combined determinations in this grouping.

[5] Standard ICP-AES scan requires 1 g to determine all of: Ag, Al, As, B, Ba, Be, Bi, Ca, Cd, Co, Cr, Cu, Fe, K, Mg, Mn, Mo, Na, Ni, P, Pb, Se, Sb, Sr, Ti, V, Zn, Zr.

[6] Hydride Vapour Atomic Absorption spectrometry. Single element determination, lower detection level than ICP-AES.

[7] Metal Mines Liquid Effluent Regulations

[8] Graphite Furnace Atomic Absorption. Useful for low levels of Cd, Cr, Ni, and Mo.

[9] Cold Vapour Atomic Absorption spectrometry

# TABLE IV: 3.4-6

Estimated weight of dried sediment layer subsampled from core liners of different diameters and approximate water content for different sediment depths.
(Modified from Mudroch and Ascue 1995)

| **Approximate weight (g) of dry material in uppermost 1 cm sediment layer**[1] | | | **Approximate**[2] **water content** in soft fine **grained sediments** | |
|---|---|---|---|---|
| **Tube Inner Diameter (cm)** | **Soft Fine-grained** Sediment[3] | Firmer **Silty Clay**[4] | **Core Depth (cm)** | Water **Content** |
| 3.5 | 0.7 - 1.4 | 1.1 - 2.2 | 10 | 80% |
| 5.08 | 1.5 - 3.0 | 2.3 - 4.6 | 20 | 70% |
| 6.6 | 2.6 - 5.2 | 3.2 - 7.9 | 30-40 | 50 - 60% |
| 10.0 | 5.9 - 11.8 | 8.8 - 17.7 | | |

[1] Based on 90 to 95 **%** sediment water content.

[2] Generality only. Will vary depending on site specific characteristics.
Sediments exceeding 90% water may extend meter's range in small lakes with high organic matter and restricted water flow.

[3] Based on specific gravity of sediment 1.5 (higher organic matter content).

[4] Based on specific gravity 2.3 (low organic matter content).

# TABLE IV: 3.4-7

Sample containers and preservation for different parameters measured in sediments.

| Parameter[a] | Container[b] | Preservation | Maximum Storage | Comments |
|---|---|---|---|---|
| Particle size | P, G or M | wet, 4°C, tightly sealed | 14 days | drying, freezing and thawing cause aggregation of particles |
| Stratigraphy | Core | wet, 4°C | several months | preserve original consistency |
| Bioassays | P or G | sieved, 4°C, dark | processed within 2 to 7 days | mixing and sieving recommended before testing sediment toxicity |
| Bacteriological | Sterile G | wet, 4°C | processed within 6 hours | |
| pH and Eh | Bucket or core | wet, undisturbed and untreated | determined in the field | very difficult and problematic temperature corrections |
| Cation Exchange Capacity | Bucket or core | wet, 4°C | determined in field | Operationally defined parameter dependent on test solution. |
| $P_{total}$, TKN | G | wet, 4°C | 1 month | if possible, analyze within 24 hours if possible |
| TOC | P or G | freeze, -20°C | 6 months, dark | carbonates and bicarbonates can interfere |
| Oil and Grease | M or G | wet, 4°C | 1 day | wet sample can be stored for up to 1 month at -10°C with 1-2 ml concentrated $H_2SO_4$ per 80 g |
| Metals | P or T | dry (60°C) or freeze (-20°C) | 6 months | if samples are not analyzed within 48 hours, freeze dried -20°C up to 6 months |
| Mercury | G or T | freeze, -20°C | 1 month | Mercury analysis is performed with wet samples |
| Volatile organics | G vials with Teflon® septums | freeze, -20°C | 1 month | no preservatives should be added; possible loss of some compounds |
| Cyanides | P | freeze, -20°C | up to 1 month | sulfide interfere colorimetry |
| Pesticides and PCB | M or G covered with Al foil | freeze, -20°C, dark | 7 days until extraction | if samples are not analyzed within 48 hours, freeze dried -20°C up to 6 months |

[a] CEC=cation exchange capacity; $P_{tot}$=total phosphorus; TNK=total Kjeldahl nitrogen; TOC=total organic carbon

[b] P=polyethylene or polypropylene; G=glass; T=Teflon®; M=metal

## TABLE IV: 3.6-1

Advantages and disadvantage of techniques commonly used for pore water sampling (from Mudroch and Ascue 1995).

| Method | Advantages | Disadvantages |
|---|---|---|
| Squeezing | Simple equipment, portable, inexpensive, sediment composition available | Oxygen contamination, $CO_2$ degassing that will change pore water composition, temperature-induced changes, pressure-related additions of metabolites |
| Centrifugation | simple, sediment composition available, easy to obtain large volumes | risks of sampling artifacts, pressure artifacts, effects of oxidation and elevated temperatures |
| Dialysis | minimal manipulation of sample, no induction of interstitial water flow, allows maximal replication, analysis of dissolved gases is possible, temperature and pressure related artifacts are avoided | disturbance of the sediment structure, need of scuba divers or submersibles, timing (minimum 12 days), risks of incomplete equilibration, risks of membrane breakdown |
| Suction | simple and easy to use, allows sampling at fairly well-defined depth | fine particles could be collected (reduction of mesh size may cause clogging), oxidation effects are hard to prevent |

# TABLE IV: 4.2-1

Erosional samplers for collecting benthic macroinvertebrates.
See TSN 4.2 for additional information.

| Sampler | Surface Area | Habitat Type | Advantages | Limitations |
|---|---|---|---|---|
| D- or Kick Net | **Qualitative**<br>Variable | **Wadeable** streams , ponds, **or** lake margins. Used to capture dislodged inverts (e.g., kicking substrate) in flowing water or as a sweep net in standing water. | Light and easy to transport. Quick means of collecting large, dominant macro-invertebrate groups. Serve as a dip net for small forage fish. | Non-quantitative. |
| Surber Sampler | Semi - **Quantitative**<br>0.1 1112 | Shallow, flowing streams, less than 32 cm in depth with good current; substrate ranging from cobble to sand. Less efficient in sand due to clogging of mesh. | Easily transportable. Sample area defined by frame. | Poor stability in strong currents. F sits on **substratel,** hence, potential **under** sampler. Water column not contained, hence, greater potential fc of organisms due to backwash cor to fully contained samplers.<br>Not efficient in still or slow movin waters. Loss of organsims increases substrate particle size increases abc mid-range for cobble. |
| Invertebrate Box Sampler | Quantitative<br>0.1 $m^2$ | As above | Sample area defined by frame. Loss beneath frame minimized as foam base conforms to most substrates. Relatively stable platform in high currents. | Not efficient in still or slow movin waters. Loss of organsims increases substrate particle size increases abc mid-range for cobble. |
| Cylinders (e.g., Hess and Neil) | Quarititative<br>0.1 $m^2$ | As above | **Sample** area defined by frame. Loss beneath frame minimized as cylinder can be twisted down into the substrate. This also stabilizes the sampler in fast currents. | Not **efficient** in still or slow movin **waters.** Loss of organsims **increase** substrate particle size increases abc mid-range for cobble. |

# TABLE IV: 4.2-I

Erosional samplers for collecting benthic macroinvertebrates.

See TSN 4.2 for additional information.

| **Sampler** | **Surface Area** | **Habitat Type** | **Advantages** | **Limitations** | **Recommended For:** |
|---|---|---|---|---|---|
| Drift Net | Semi-Quantitative<br>Volumetric sampler | All substrate types. Samples drifting organisms in the Water column. Collects relatively quantitative samples on volumetric basis (e.g., area of sample mouth*time in place*current velocity). | Low sampling error. Less expenditure of personnel time in actually sampling , or washing and sorting of invertebrates. May be used in all substrates. | Low sampling error off-set when converting to a volumetric basis due to fluctuations in discharge, diurnal drift patterns, and clogging of the mesh. Hence, errors increase with time in place.<br>Not known where organisms were when they originally entered the drift. Does not sample non-drifting organisms. | Not recommended for routine benthic community investigtions. Of use in special investigations of water contaminant pulse or "slug" releases |

[1] Base extensions are available to minimize this problem.

Samplers vary in size depending on the manufacturer, therefore it is important to always confirm the dimensions of the sampler used. Mesh size is also important.

## TABLE IV: 4.2-2

Recommended level of taxonomic precision in benthic macroinvertebrate sample analysis.*

| Group | Level |
|---|---|
| Coelenterata | genus |
| Turbellaria | family |
| Nematoda | phylum |
| Nemertea | genus |
| Polychaeta | species |
| Oligochaeta: | |
| . Tubificidae | species (except some immatures) |
| . Naididae | species |
| . Lumbriculidae | species |
| . Other families | genus |
| Hirudinea | species |
| Bivalvia: | |
| . Pisidiidae | Genus (Pisidium), species (Sphaerium) |
| . Unonidae | species |
| Gastropoda | species |
| Crustacea: | |
| . Ostracoda | order |
| . Harpacticoida | subclass |
| . Others (benthic only) | species |
| Acarina | order |
| Insecta | genus |

*Table originally provided in the Environmental Effects Monitoring Technical Guidance Document (EC and DFO 1993).

# TABLE IV: 5.2-1

Description of variables used in the health assessment index (HAI). Values are assigned to each of these variables according to the type and severity of each observed anomaly

| Variable | Variable Condition | [1]Original Field Designation | [2]Substituted Value For The HAI |
|---|---|---|---|
| Thymus | No hemorrhage | 0 | 0 |
| | Mild hemorrhage | 1 | 10 |
| | Moderate hemorrhage | 2 | 20 |
| | Severe hemorrhage | 3 | 30 |
| Fins | No active erosion | 0 | 0 |
| | Light Active erosion | 1 | 10 |
| | Moderate active erosion with some hemorrhaging | 2 | 20 |
| | Severe active erosion with hemorrhaging | 3 | 30 |
| Spleen | Normal; black, very dark red, or red | B | 0 |
| | Normal; granular, rough appearance of spleen | G | 0 |
| | Nodular, containing fistulas or nodules of varying size | D | 30 |
| | Enlarged; noticeably enlarged | E | 30 |
| | Other; gross aberrations not fitting above categories | OT | 30 |
| Hindgut | Normal; no inflammation or reddening | 0 | 0 |
| | Slight inflammation or reddening | 1 | 10 |
| | Moderate inflammation or reddening | 2 | 20 |
| | Severe inflammation or reddening | 3 | 30 |
| Kidney | Normal firm dark red color, lying relatively flat along the length of the vertebral column | N | 0 |
| | Swollen; enlarged or swollen wholly or in part | S | 30 |
| | Mottled; gray discoloration | M | 30 |
| | Granular; granular appearance and texture | G | 30 |
| | Urolithiasis or nephrocalcinosis; white or cream-colored mineral material in kidney tubules | U | 30 |
| | Other; any aberrations not fitting previous categories | OT | 30 |
| Skin | Normal; no aberrations | 0 | 0 |
| | Mild skin aberrations | 1 | 10 |
| | Moderate skin aberrations | 2 | 20 |
| | Severe skin aberrations | 3 | 30 |
| Liver | Normal; solid red or light red color | A | 0 |
| | "Fatty" liver, "coffee with cream" color | C | 30 |
| | Nodules in the liver; cysts or nodules | D | 30 |
| | Focal discoloration; distinct localized color changes | E | 30 |
| | General discoloration; color change in whole liver | F | 30 |
| | Other; deviation in liver not fitting other categories | OT | 30 |

# TABLE IV: 5.2-1

Description of variables used in the health assessment index (HAI). Values are assigned to each of these variables according to the type and severity of each observed anomaly

| Variable | Variable Condition | [1]Original Field Designation | [2]Substituted Value For The HAI |
|---|---|---|---|
| Eyes | No aberrations: good "clear" eye | N | 0 |
| | Generally; an opaque eye (one or both) | B | 30 |
| | Swollen; protruding eye (one or both) | E | 30 |
| | Hemorrhaging or bleeding in the eye (one or both) | H | 30 |
| | Missing one or both eyes | M | 30 |
| | Other: any manifestation not fitting the above | OT | 30 |
| Gills | Normal; no apparent aberrations | N | 0 |
| | Frayed; erosion of tips of gill lamellae resulting in "ragged" gills | F | 30 |
| | Clubbed; swelling of the tips of the gill lamellae | C | 30 |
| | Marginate; gills with light, discolored margin along tips of the lamellae | M | 30 |
| | Pale; very light in color | P | 30 |
| | Other, any observation not fitting above | OT | 30 |
| Pseudobranchs | Normal; flat containing no aberrations | N | 0 |
| | Swollen; convex in aspect | S | 30 |
| | Lithic; mineral deposits. white, somewhat amorphous spots | L | 30 |
| | Swollen and lithic | S&L | 30 |
| | Inflamed; redness, hemorrhage, or other | I | 30 |
| | Other; any condition not covered above | OT | 30 |
| Parasites[3] | No observed parasites | 0 | 0 |
| | Few observed parasites | 1 | 10 |
| | Moderate parasite infestation | 2 | 20 |
| | Numerous parasites | 3 | 30 |
| The following parameters require quantififcation for specific fish species[4]. | | | |
| Hematocrit | Normal range | | |
| | Above normal range | | |
| | Below normal range | | |
| | Below normal range | | |
| Leukocrit | Range defined as normal | | |
| | Outside the normal range | | |
| Plasma protein | Normal range | | |
| | Above normal range | | |
| | Below normal range | | |

---

[1] Goede and Barton 1990

[2] Adams et al. 1993

[3] Note comments in text (Ch. IV: 5.2.1.2) regarding the assumption that absence of parasites is positive health sign.

[4] Adams et al. 1993 provides values for centrarchid species such as largemouth bass and redbreast sunfish.

## TABLE IV: 5.3-1

Life history characteristics[1] for species reported in the Lesser Slave River, Alberta.

| Species | Scientific Name | Food | Spawning Time | Fecundity | Growth Rate[2] | Longevity (years) | Age to Maturation (years) | Abundance | Ranking |
|---|---|---|---|---|---|---|---|---|---|
| Mountain whitefish | *Prosopium williamsoni* | benthos | fall | < 10000 | 55% | <10 | 3 | high | +1 |
| Lake whitefish | *Coregonus clupeaformis* | benthos/plankton | fall | c 10000 | 55% | >20 | 7 | high | +1 |
| Cisco | *Coregonus artedii* | benthos | fall | >20000 | 40% | <10 | 2-4 | low | -1 |
| Arctic grayling | *Thymallus arcricus* | terrestrial insects | spring | <7000 | 40% | <9 | 4 | low | -3 |
| Northern pike | Esox *lucius* | fish | spring | >30000 | 100% | >20 | 4-6 | high | +5 |
| Walleye | *Stizostedion vitreum* | fish | spring | >50000 | 65% | <20 | 4 | low | +3 |
| Burbot | *Lota lota* | fish | mid-winter | >50000 | 55% | >10 | 3-4 | low | +1 |
| White sucker | *Cutastomus* commersoni | benthos | spring | >20000 | 50% | <15 | 4-8 | high | +7 |
| Longnose sucker | *Cutastomus catostomus* | benthos | spring | >20000 | 100% | <15 | 5-7 | high | +7 |
| Goldeye | *Hiodon alosoides* | fish/insects | late spring/summer | >20000 | 25% | <15 | 6-10 | low | -3 |
| Perch | *Perca flavescens* | benthos/fish | late spring/summer | >30000 | 85% | <10 | 3-4 | high | +1 |

[1] From Munkittrick 1992

[2] Growth rates were evaluated as the average change in body size during the period from 3 to 7 years of age for that particular species. Since this age range spans the age of maturation for most species, species are ranked higher if they have a high energetic demand for somatic growth during the time of maturity and first spawning.

[3] Desired characteristics were given +1 and undesired -1, maximum ranking +7. This particular design was optimized for spring spawners because of numerous tributary spawning areas near the proposed outfall.

# TABLE IV:5.3-2

Definition of terms used to describe the spawning condition (gonad maturation) of fish.

| **Term** | **Definition** |
|---|---|
| Immature | Gonads of juvenile fish are generally very small and threadlike or stringlike and are just beneath the vertebral column. They appear transparent or translucent. In some species the ovaries are wine-red and the testes are whitish to grey-brown |
| Green (Maturing) | As the gonads develop they gradually enlarge until they fill the body cavity. When mature the testes are usually white. Mature ovaries become opaque and are yellow or orange. Eggs or milt are not released when pressure is applied to the abdomen. |
| **Ripe** | Eggs or milt are released with pressure applied to the abdomen. |
| Ripe and Running | Eggs or milt are released with little or no application of pressure. |
| Spent | The gonads have been emptied of the reproductive products. Abdomen of adult fish (particularly females) is soft. Some milt or eggs may remain. |

# TABLE IV: 5.3-3

Anatomical structures commonly used for age determination.
(from Env. Can. and DFO 1993)

| Family | Common Name | Structure |
|---|---|---|
| Acipenseridae | Sturgeon | Otolitbs, pectoral fin ray |
| Anguillidae | Freshwater eels | Otoliths |
| Salmonidae | Trouts | Scales, otolitbs, subopercula, 1st four marginal pectoral fin rays |
| Esocidae | Pikes | Scales, cleithrae |
| Cyprinidae | Minnows and carps | Scales |
| Catostomidae | Suckers | Scales, fin rays, opercular bone |
| Ictaluridae | Bullhead catfishes | Pectoral spines |
| Cyprinodontidae | Killifishes | Scales |
| Atherinidae | Silversides | Otoliths |
| Percichtbyidae | Temperate basses | Scales |
| Centrarchidae | Sunfishes, bass | Scales, otolitbs, 1st three dorsal spines, opercular bone |
| Sciaenidae | Drums | Scales |
| Percidae | Walleye, perch | Otoliths, 1 st three dorsal spines, opercular bone, scales |
| Coregoninae | Whitefish | Otoliths, 1st four marginal pectoral fin rays, scales |

# TABLE IV: 5.3-4

Sample checklist for physical examination of fishes'.

**Date:** ____________________

Location: ____________________

Control no: ____________________

Sampling method: ____________________

Recorder: ____________________

Species: ____________________

| | | |
|---|---|---|
| Length (mm) | Standard: | __________ |
| | **Total:** | __________ |
| | Fork | __________ |
| Weight(g) | Fresh: | __________ |
| | Gonad: | __________ |
| | Liver: | __________ |

Sex: ____________________

Tissue samples: ____________________

**BODYFORM**

**Normal** __________

Emaciated __________

Truncate __________

scoliosus __________

Lordosis __________

**BODY SURFACE**

**Normal** __________

Raised scales __________

Swollen __________

Lesions __________

Excess mucus __________

Reoriented scales __________

**Growths** __________

Parasites __________

Wounds (lamprey) __________

**UPS AND JAWS**

**Normal** __________

Deformed __________

Growths __________

**SNOUT**

**Normal** __________

**Pugnose** (pughead) __________

**ISTHMUS**

**Normal** __________

Enlarged __________

Hemorrhagic __________

**EYES**

**Normal** __________

**Popeye** __________

Cloudy cornea __________

Missing __________

Lens deformed __________

Lens parasites __________

Lens cataracts __________

**FINS**

**Normal** __________

Frayed-eroded __________

Parasites __________

Hemorrhagic __________

Gas bubbles __________

---

[1] Adapted from Hunn, 1988

Page 1 of 2

**FINS-ERODED**

Dorsal ____________________

Pectoral ____________________

Pelvic ____________________

Anal ____________________

**BRANCHIAL CAVITY**

Normal ____________________

Growths ____________________

Parasites ____________________

**LESIONS-location**

Fins ____________________

Head ____________________

Eyes ____________________

Mouth ____________________

Peducle ____________________

Ventral ____________________

Dorsal ____________________

Lateral ____________________

**BARBELS** (if present)

Normal ____________________

Deformed ____________________

Missing ____________________

**GILLS**

Normal ____________________

Bright Red ____________________

Brown ____________________

Gas Bubbles ____________________

Parasites ____________________

**OPERCULUM**

Normal ____________________

Incomplete ____________________

**SEXUAL CONDITION**

Gravid ____________________

## References

Hunn, J.B. 1988. Field assessment of the effect of contaminants on fishes. U.S. Department of the Interior, Fish and Wildlife Service Biological Report 88 (19). 27 pp.

Page 2 of 2

# TABLE IV: 5.3-5

Recommended fish collection gear (adapted from Environment Canada, 1992b).

| Sampler | Habitat | Effectiveness | Advantages | Limitations |
|---|---|---|---|---|
| Bectroshocker | Flowing waters, such as ivers/streams, fresh and saltwater channels, and lake shores | Useful for collections in elatively shallow, clear, flowing waters | Portable (backpack) and boat-mounted types used; access to difficult locations, narrow streams/rivers | Efficiency decreases in water with very low or very high specific conductivities; poor in nrbid waters |
| Gill Net | 411 habitats | Useful for size-selective collections; can be used n shallow and deep waters | Can be deployed with (deeper waters) or without (shallow waters) a boat; low effort; variable mesh size for size-selection | Fish may not remain viable for long; net can rap non-target organisms, such as fish-eating bids |
| Seine Net | All habitats | Useful in shallow water; most useful for collecting smaller fish | Easily deployed by two people in shallow waters | Easily fouled on rocks or other debris |
| Purse Seine | All habitats except fast-flowing rivers. and small rivers/streams | Used to encircle pelagic fish in deep waters; useful for collecting larger fish | **Relatively large areas can be encircled; large sample sizes can be obtained** | Boat is needed to deploy |
| Otter Trawl | Lakes, estuaries, oceans | Used mainly to collect benthic fish | **Useful in shallow and** deep waters | Boat is needed to deploy |
| Angling | All habitats | Useful in sites difficult to access with gear (eg., under ice, fast current) | Useful in shallow and deep waters | Small sample sizes are collected; biased in term of species caught, age, size and condition of fis |
| Trap Nets,<br>Hoop Nets,<br>Fyke Nets | All habitats but open ocean | Generally not size selective; best suited to shallow water | Useful for catching fish in difficult to sample locations and at awkward times (eg, night); low effort | Trap will be biased to th more mobile species and age classes; baited traps (minnow traps) may be biased by the selected bait |
| Piscicide (e.g., **Rotenone®**) | All habitats but open ocean | Useful only in backwaters, of limited use | Effective in killing broad range of fish | Fish are killed; may not kill all fish, especially **benthic** species |

**FINS-ERODED**

Dorsal ____________________
Pectoral ____________________
Pelvic ____________________
Anal ____________________

**BRANCHIAL CAVITY**

Normal ____________________
Growths ____________________
Parasites ____________________

**LESIONS-location**

Fins ____________________
Head ____________________
Eyes ____________________
Mouth ____________________
Peduce ____________________
Ventral ____________________
Dorsal ____________________
Lateral ____________________

**BARBELS** (if present)

Normal ____________________
Deformed ____________________
Missing ____________________

**GILLS**

Normal ____________________
Bright Red ____________________
Brown ____________________
Gas Bubbles ____________________
Parasites ____________________

**OPERCULUM**

Normal ____________________
Incomplete ____________________

**SEXUAL CONDITION**

Gravid ____________________

## References

Hunn, J.B. 1988. Field assessment of the effect of contaminants on fishes. U.S. Department of the Interior, Fish and Wildlife Service Biological Report 88 (19). 27 pp.

# TABLE IV: 5.3-6

Relative changes in population characteristics associated with the various response patterns.
(Based on Munkittrick 1989a)

| Response Pattern | Mean/ Age | Age/ Distribution | Growth Rate | Condition Factor | Age at Maturation | Fecundity | Egg Size | Population Size |
|---|---|---|---|---|---|---|---|---|
| **Type 1** Exploitation |  | shift to younger | + | + |  | + | -/+ | - |
| **Type 2** Recruitment Failure | + | shift to older | 0/+ | 0 | 0 | 0 | 0 | - |
| **Type 3** Multiple Stressors | + | shift to older |  |  | + | - | -/0 | - |
| **Type 4** Limitations | 0 | 0 |  |  | + |  |  | + |
| **Type 5** Niche Shift | 0 | 0 |  |  | + |  |  | -/0 |

## LIST OF FIGURES

### CHAPTER I

### CHAPTER II

### CHAPTER III

## CHAPTER IV

**APPENDIX I - TECHNICAL SUMMARY NOTES** *(separate document)*

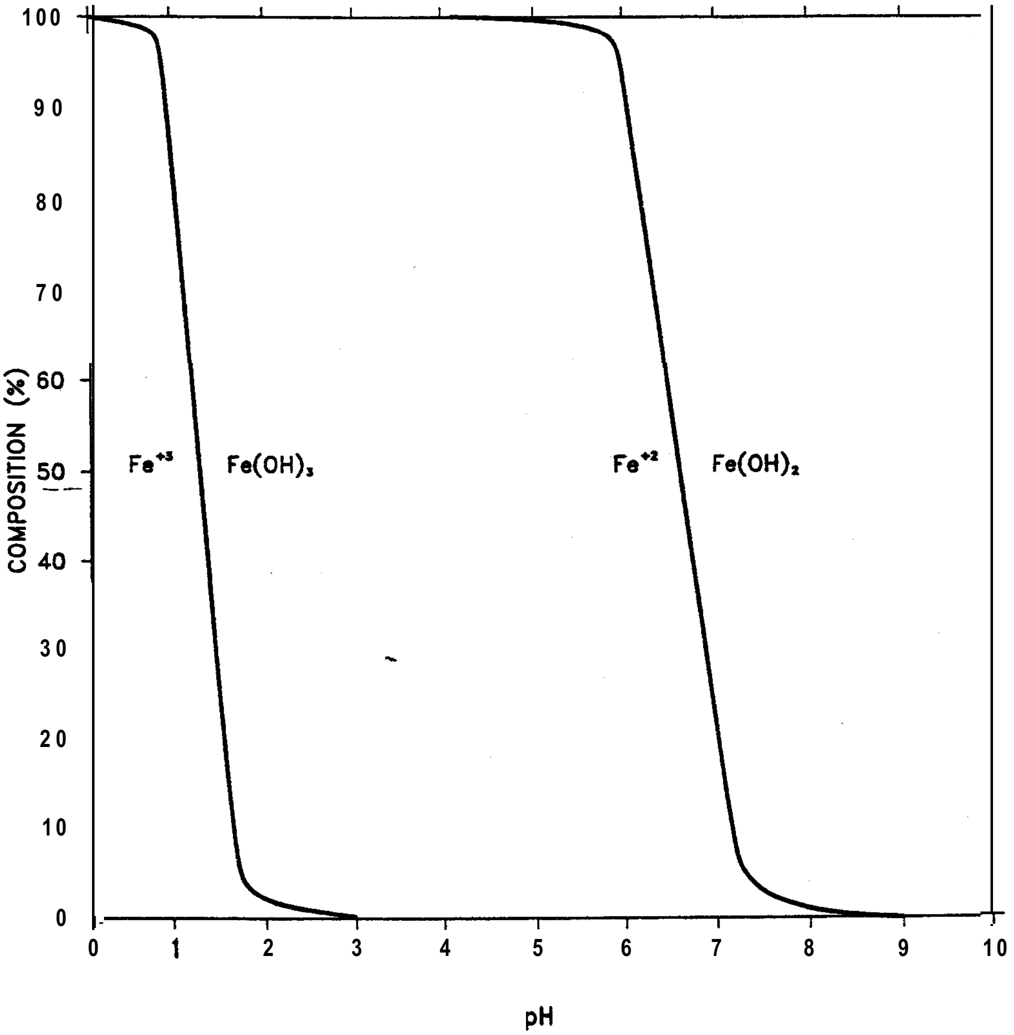


FIGURE II: 1.1-1
Distribution of ferrous and ferric iron as a function of pH
(from Ritcey 1989a).

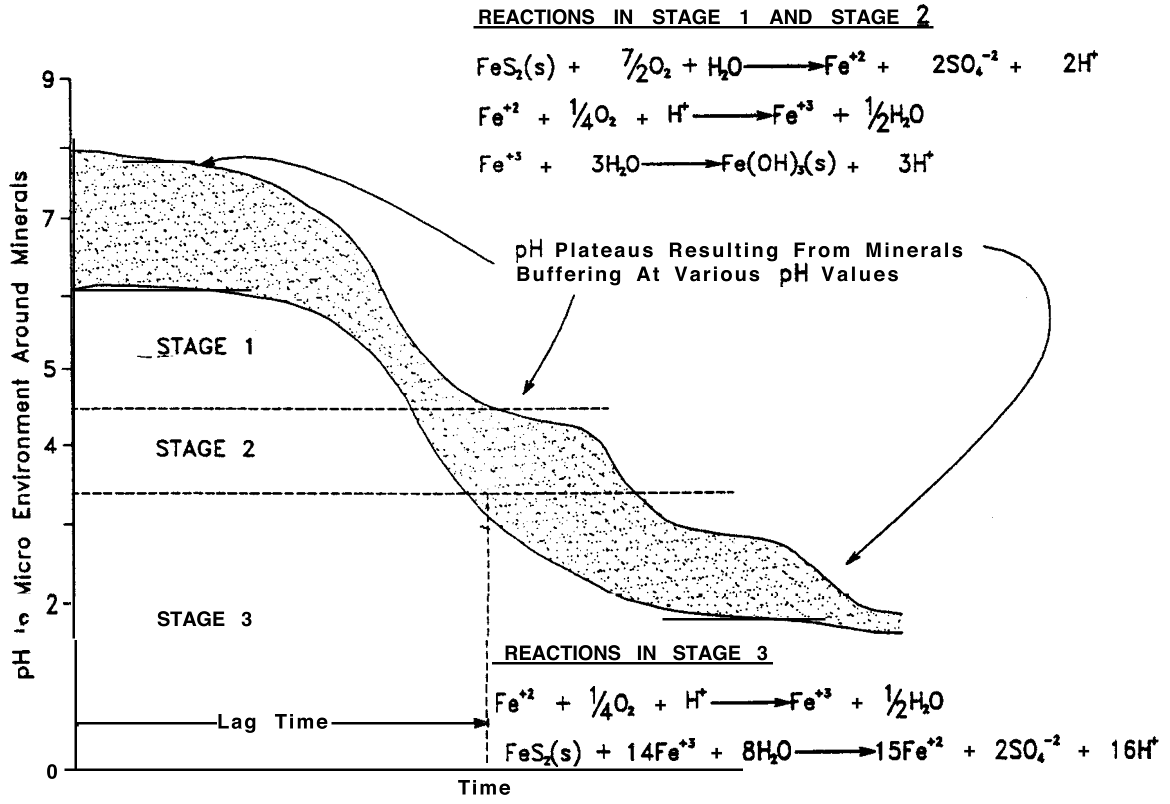


FIGURE II: 1.2-1
The three conceptual stages in the formation of an acid mine drainage (BC AMD 1989a).

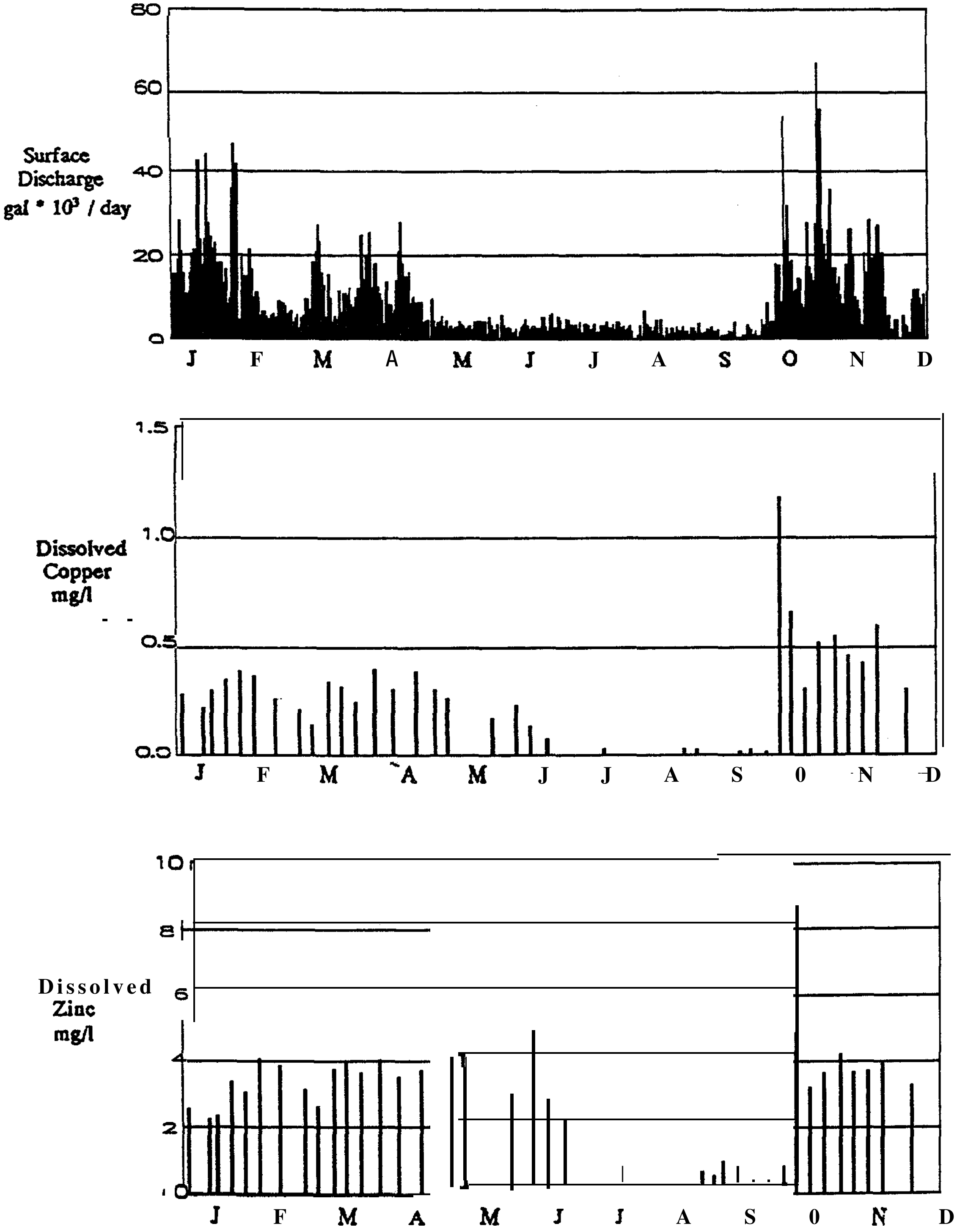


FIGURE II: 1.4-1
Seasonal patterns of flow and metal-concentrations at a coastal mine (modified from BC AMD 1990).

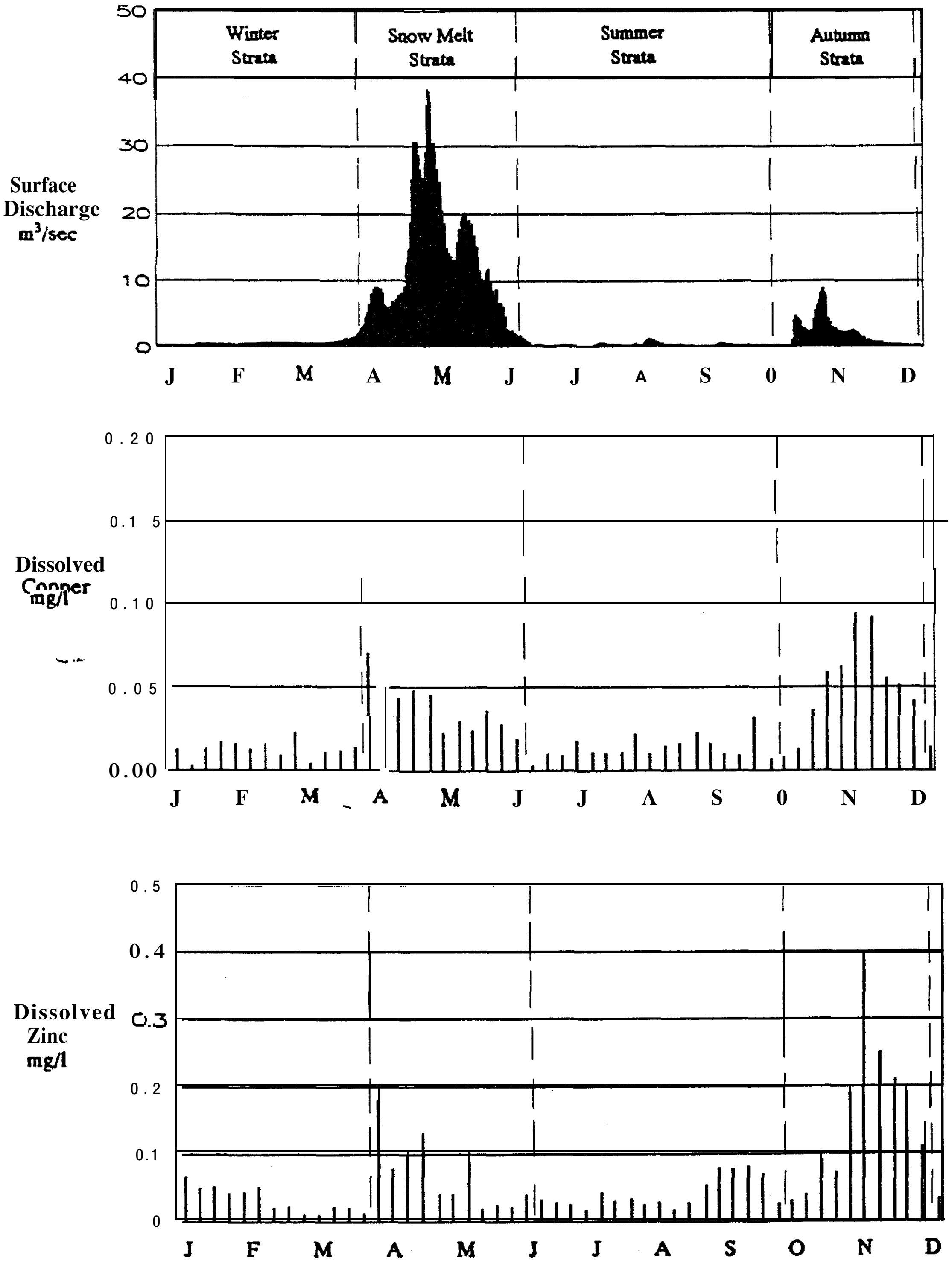


FIGURE II: 1.4-Z
Seasonal patterns of flow and metal-concentrations at an interior mine (modified from BC AMD 1990).

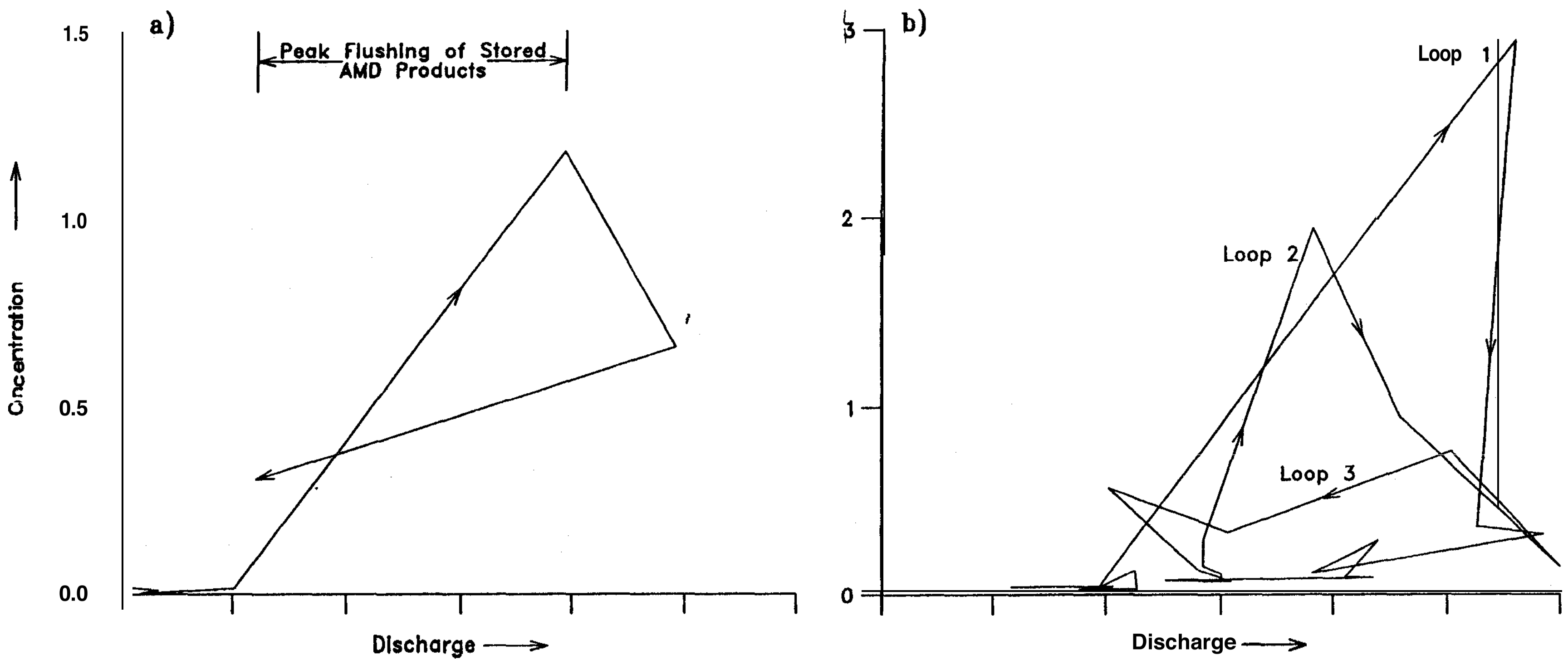


FIGURE II: 1.4-3
Hysteresis effect at two different mine sites. a) a single hysteresis loop for one week of dissolved copper data from the first fall rain; b) several diminishing hysteresis loops for one month of zinc concentrations (modified from BC AMD 1990b).

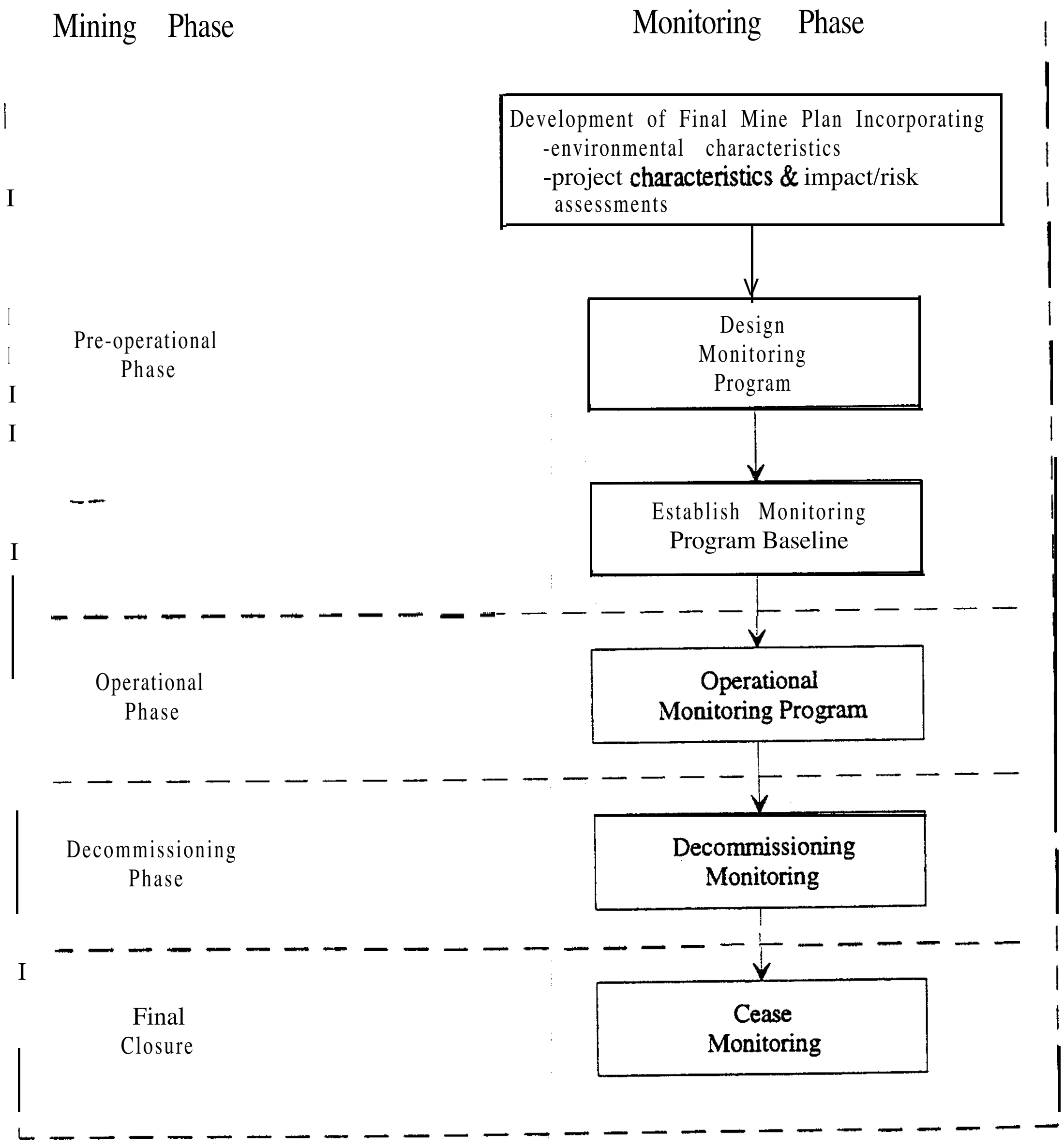


FIGURE II: 2.1-1
The corresponding phases of mine status and the monitoring program.

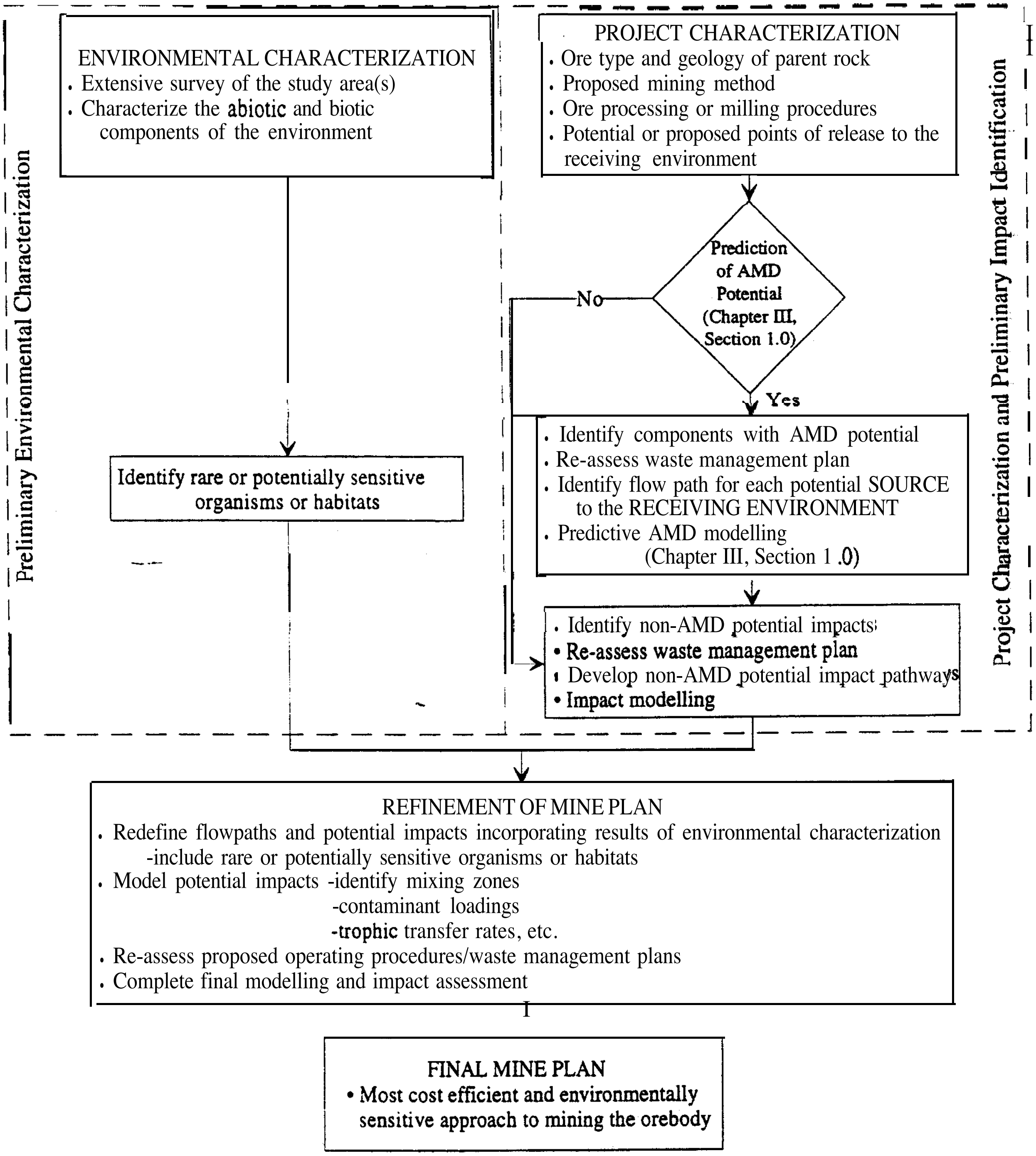


FIGURE II: 2.2-1
Framework for the development of the final mine plan.

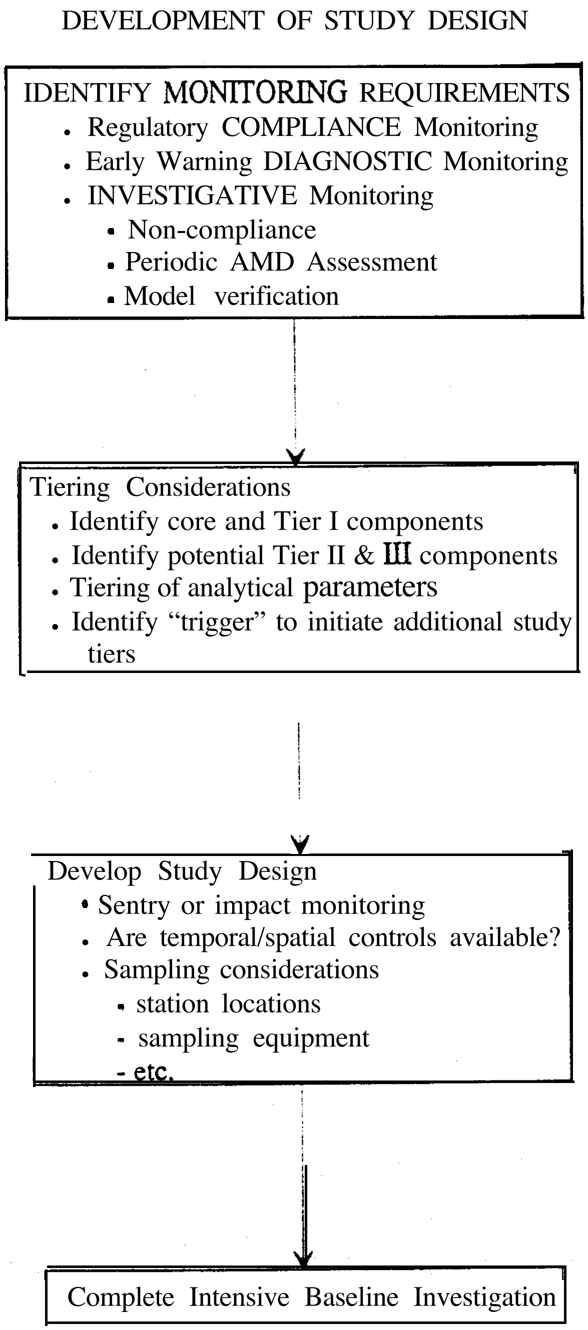


FIGURE II: 2.2-2
Framework for the development of the study design for the operational monitoring program.

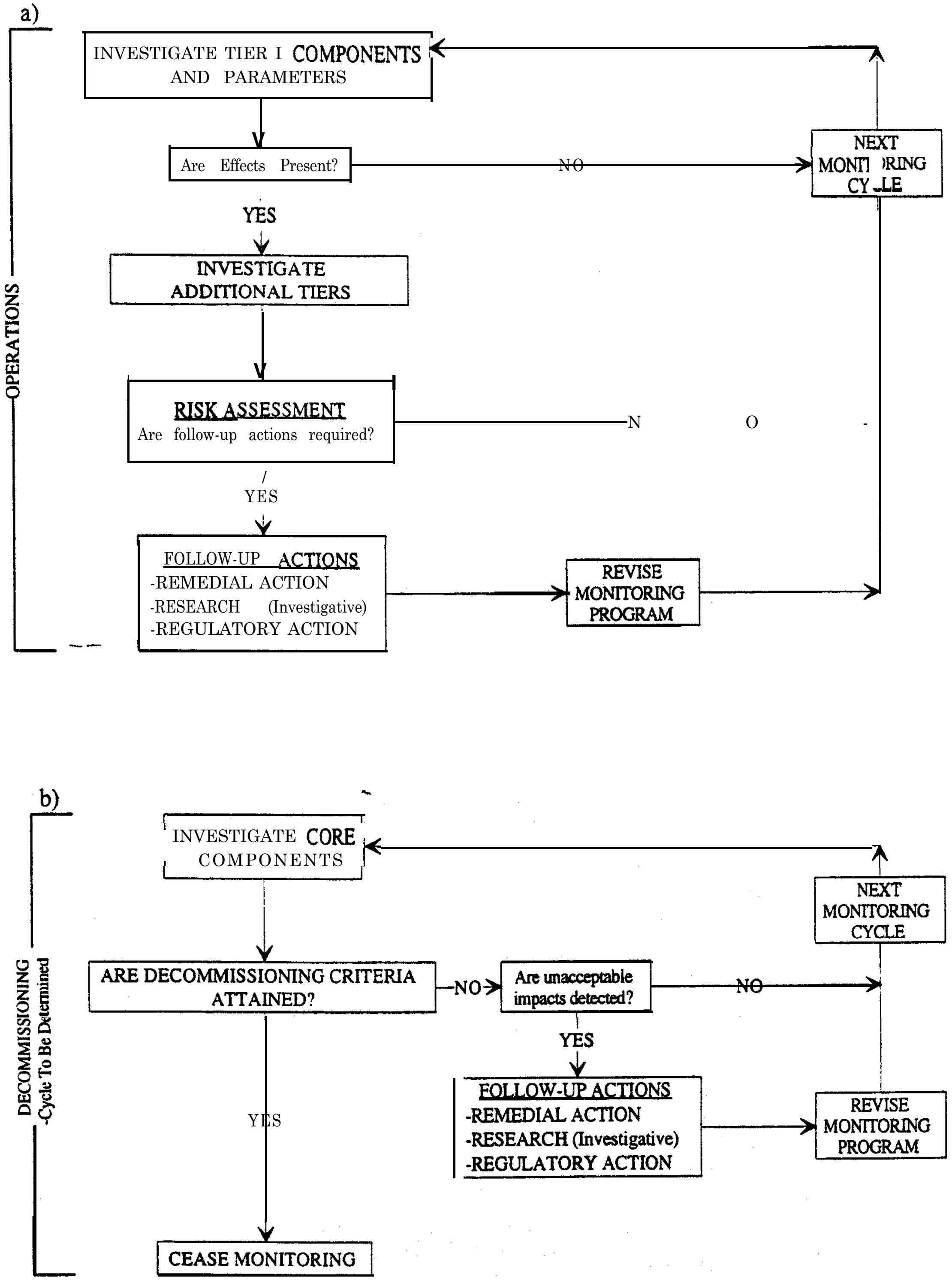


**FIGURE** II: 2.3-1
Schematic flow design for a) operational and b) decommissioning monitoring programs.

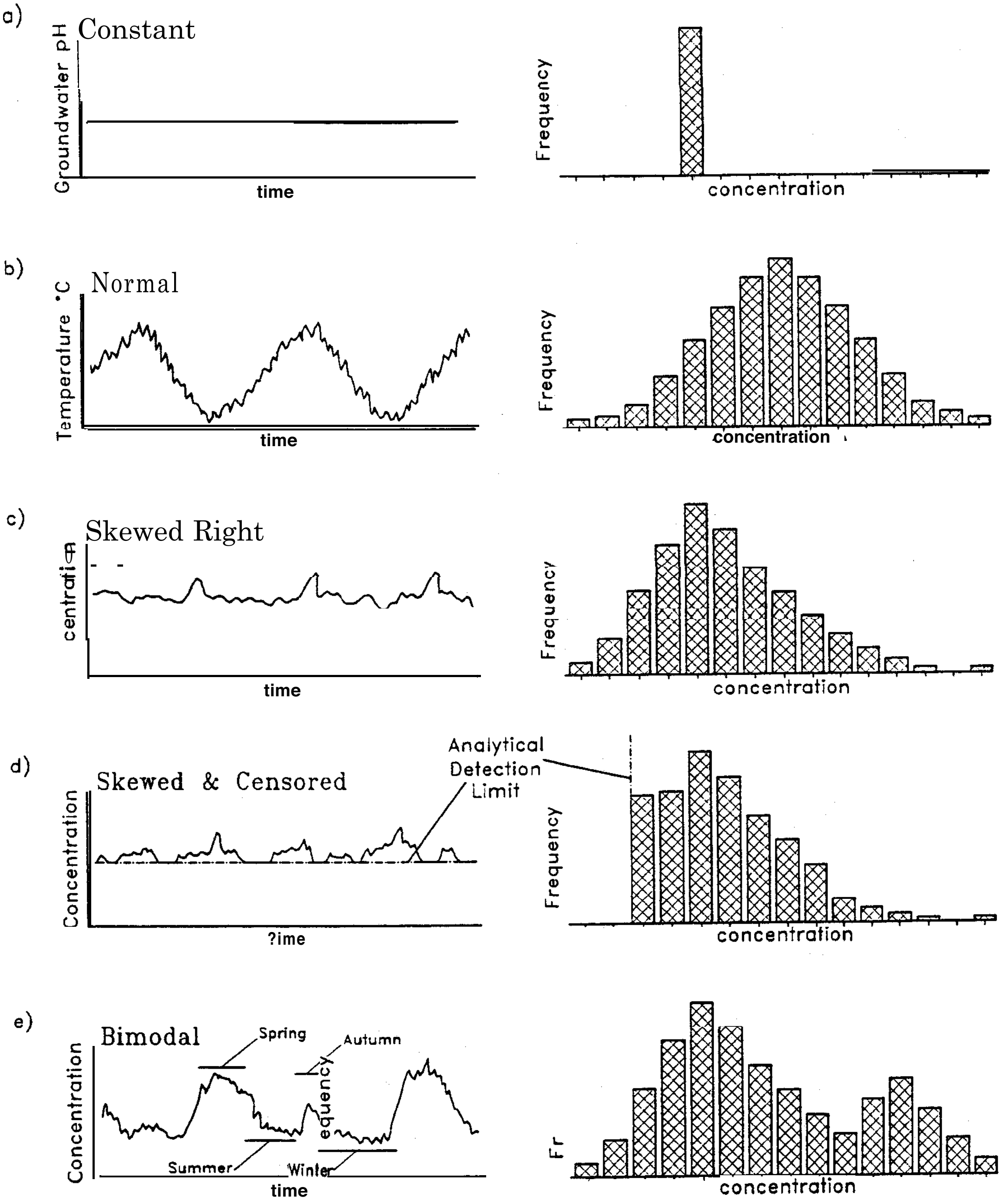


FIGURE II: 2.5-1
Examples of concentration changes over time and their corresponding frequency distributions. Figures show conceptual patterns and are not based on real data (digitized from BC AMD 1990).

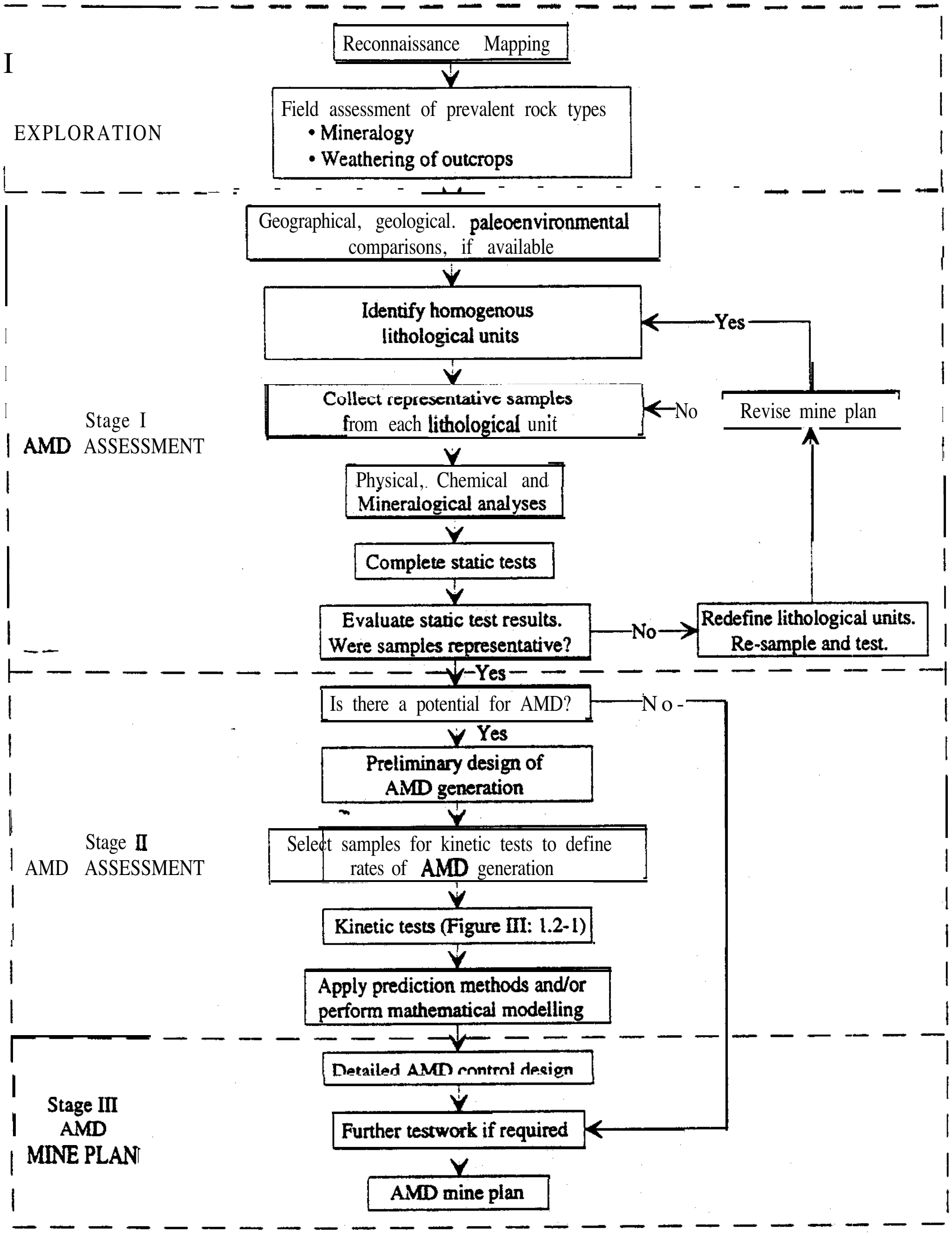


**FIGURE III:** 1.1-1
Procedure for evaluating AMD potential (BC AMD 1989a)

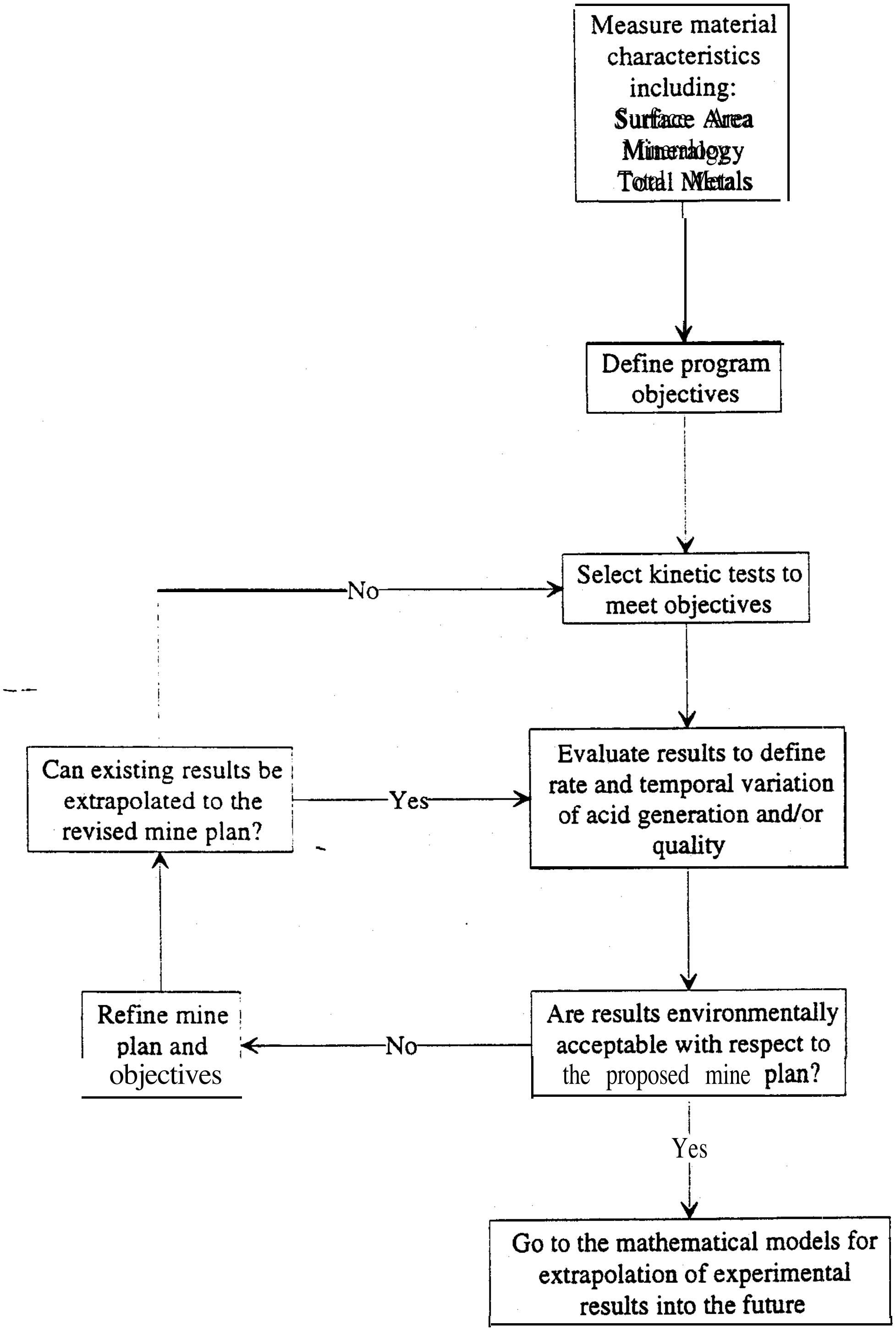


FIGURE III: 1.2- 1
Recommended kinetic test procedure (BC AMD 1989a).

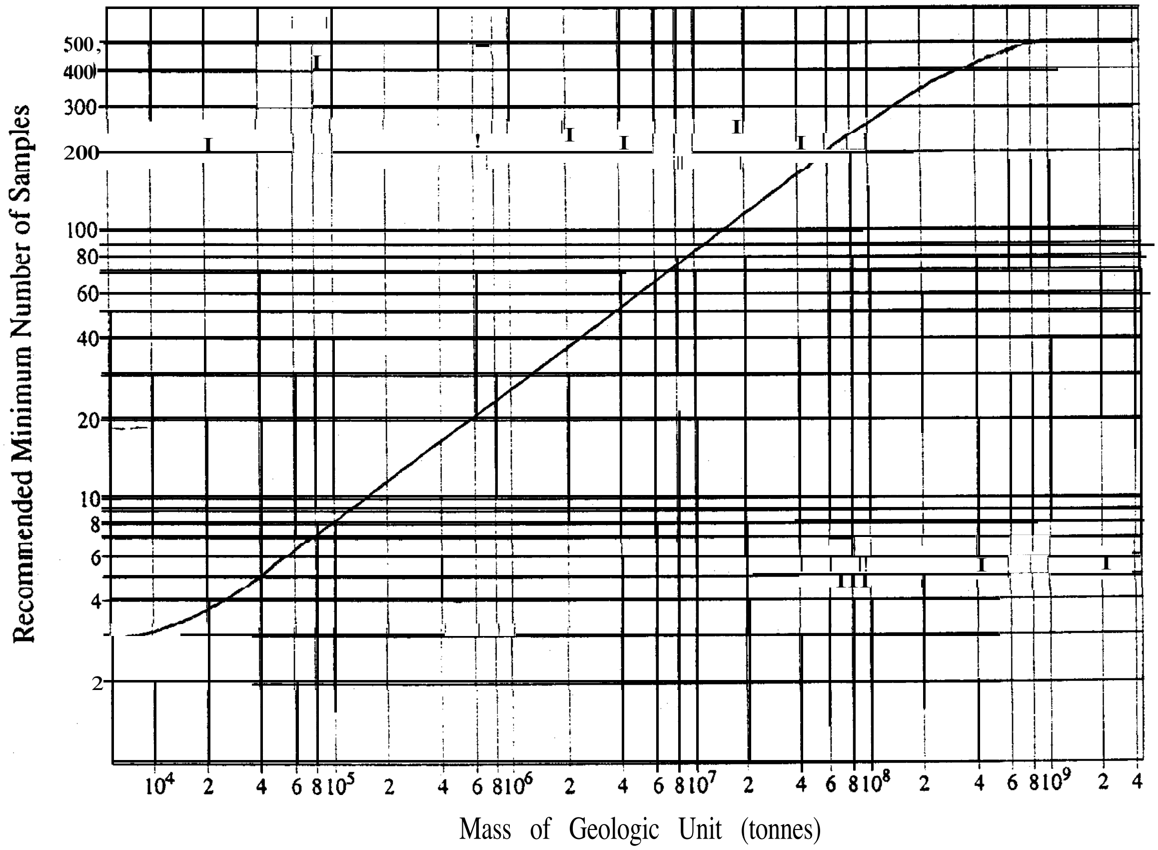


FIGURE III: 1.3-1

Recommended minimum number of samples as a function of mass of each geologic unit (BC AMD 1989a).

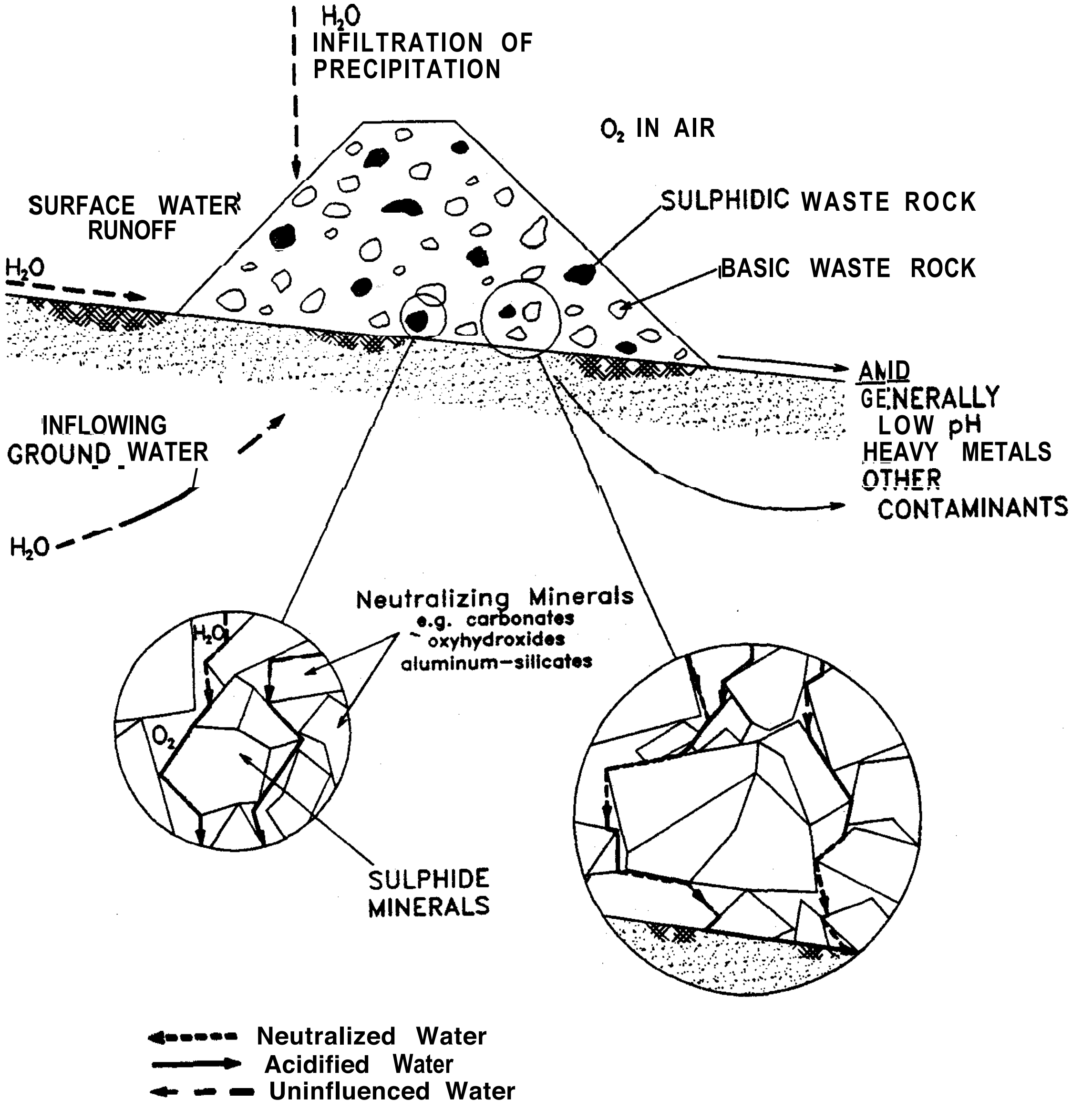


FIGURE III: 2.2-1
A conceptual diagram of acid generation and AMD migration in a waste rock pile (from BC AMD 1989a).

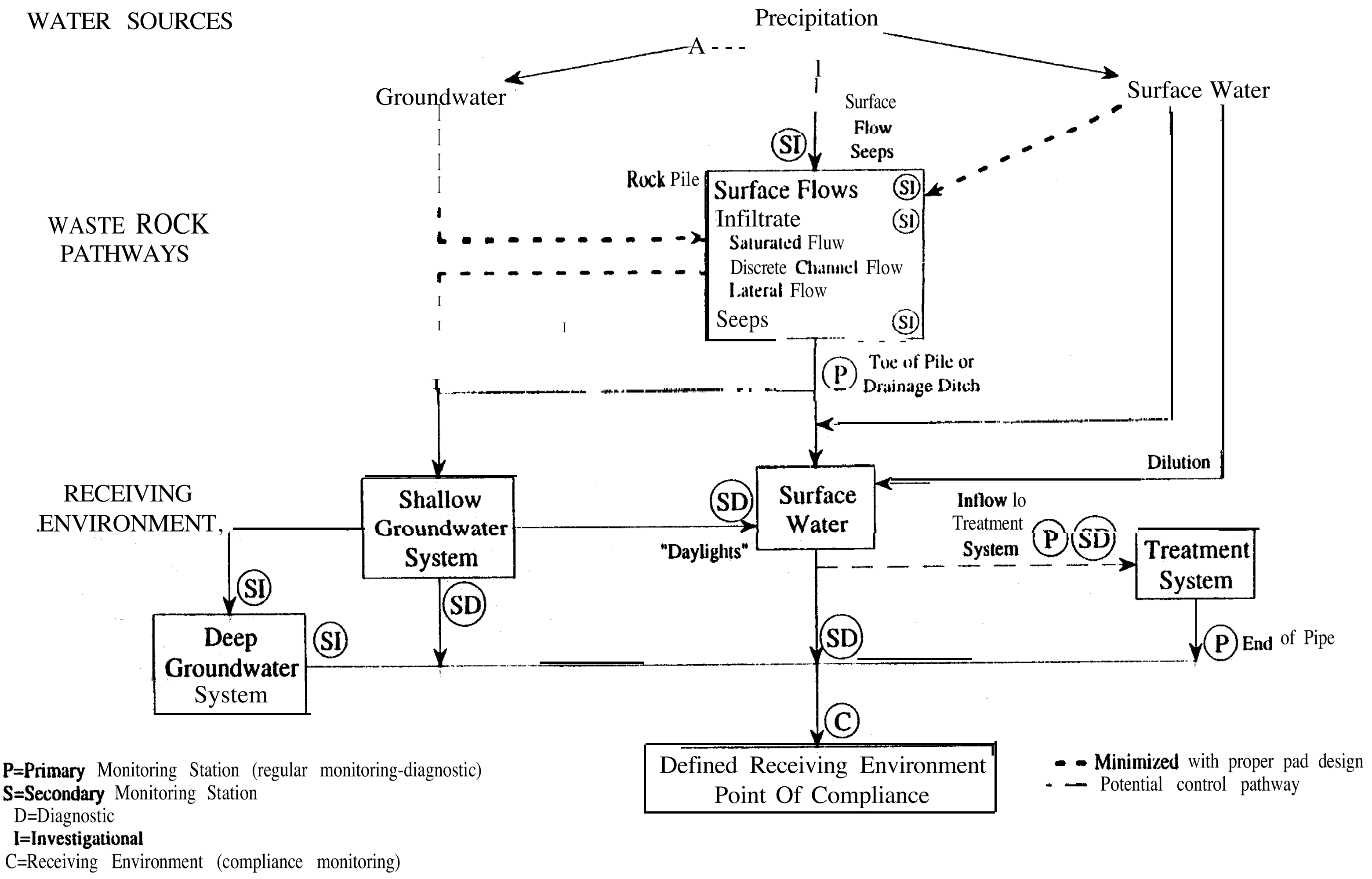


FIGURE 111: 2.2-2 Sources and pathways of AMD in a rock pile.

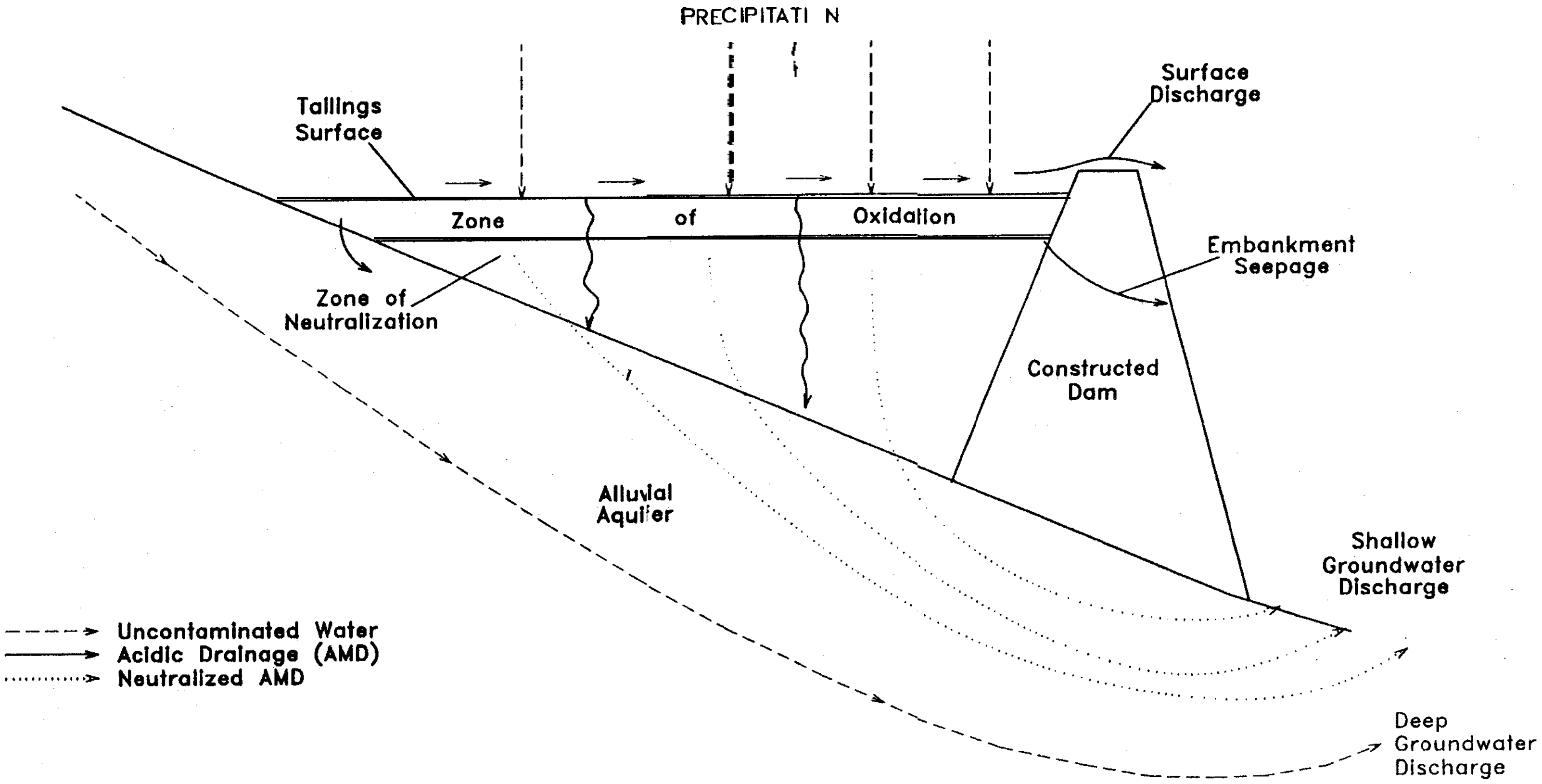
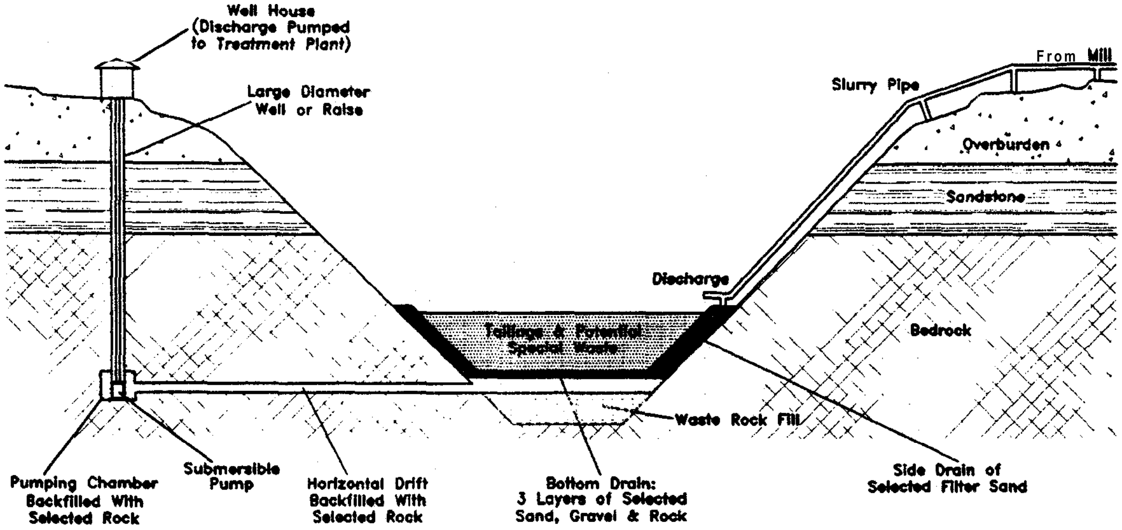


FIGURE III: 2.3-1
Conceptual illustration of the AMD process in tailings impoundment.

Stage I



Stage II

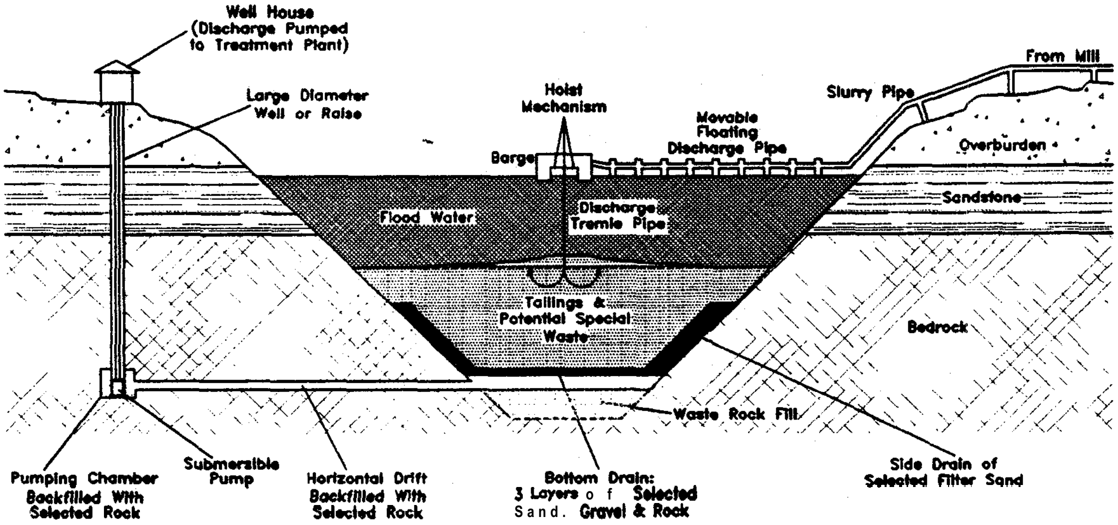


FIGURE III: 2.3-2
Tailings management facility for the Key Lake Uranium Mine in northern Saskatchewan. Facility uses the mined out Dielmann Pit & incorporates three developmental stages, the first two shown here.
Stage I) Sub-aerial deposition with bottom and side drains to optimize consolidation of the tailings. Stage II) Shift to subaqueous deposition. Flooding occurs when dewatering activities cease. Stage III) (not shown) involves the deposition of a subaqueous cap consisting of inert coarse- and fine-grained soils. (Cameco 1994).

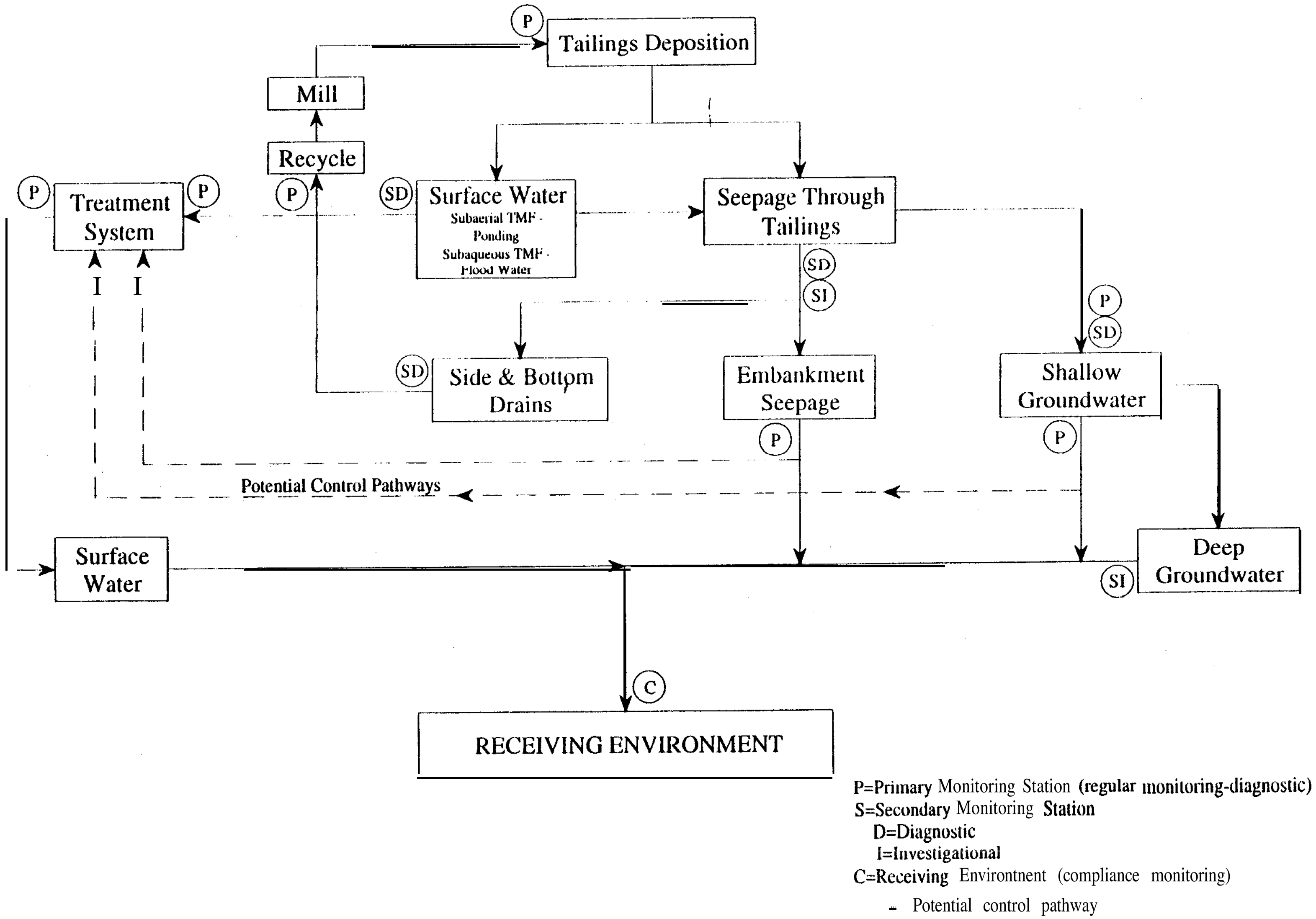
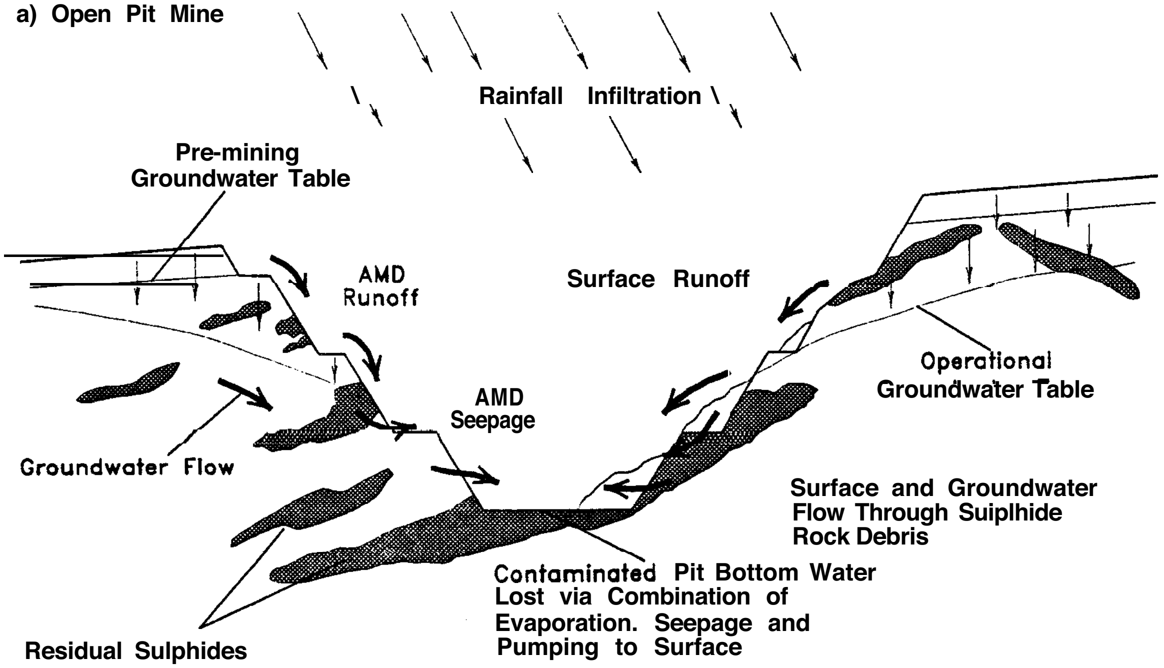


FIGURE III: 2.3-3 Sources and pathways of AMD in tailings.

a) Open Pit Mine



b) Underground Mine

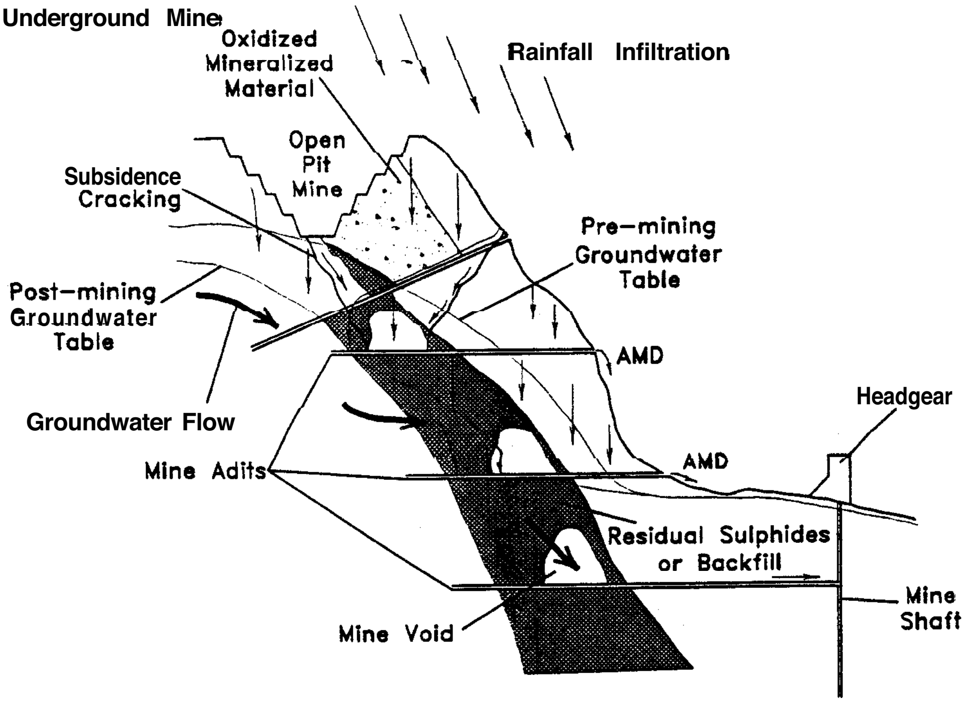


FIGURE III: 2.4-1
Conceptual illustration of the **AMD** process in a) an open pit mine and b) an underground mine.

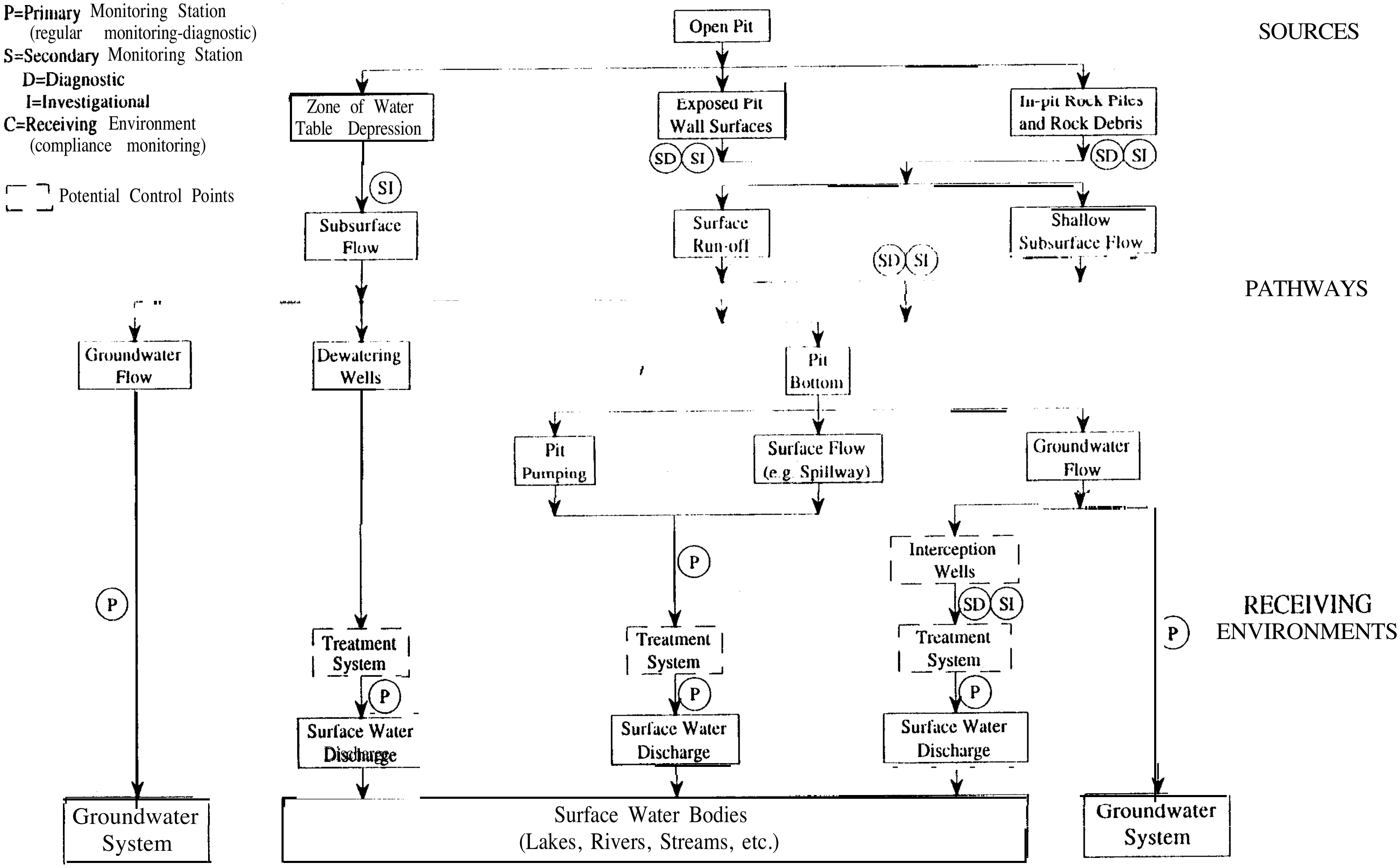


FIGURE III: 2.4-2 Sources and pathways of AMD in an open pit mine.

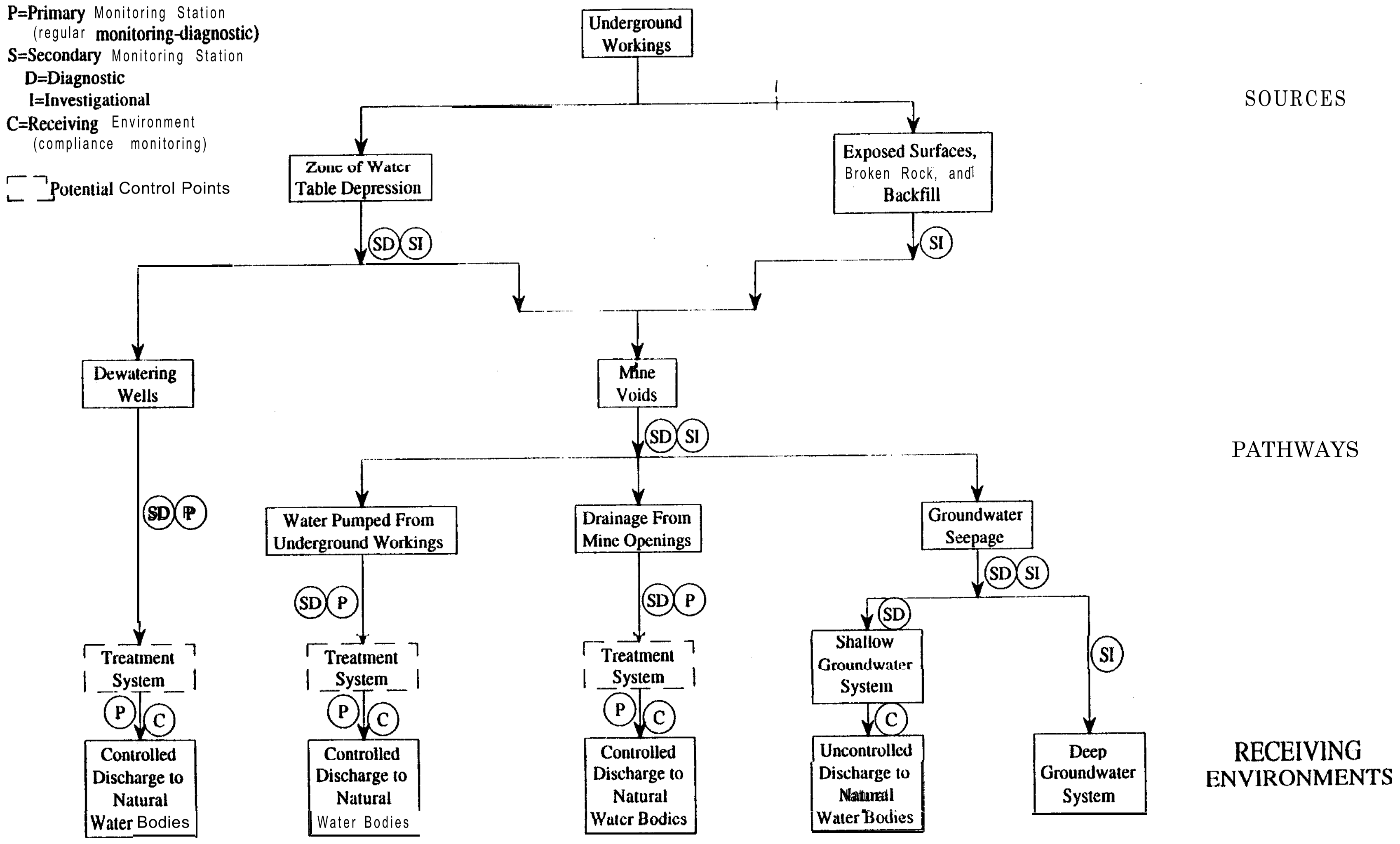


FIGURE III: 2.4-3 Sources and pathways of AMD in an underground mine.

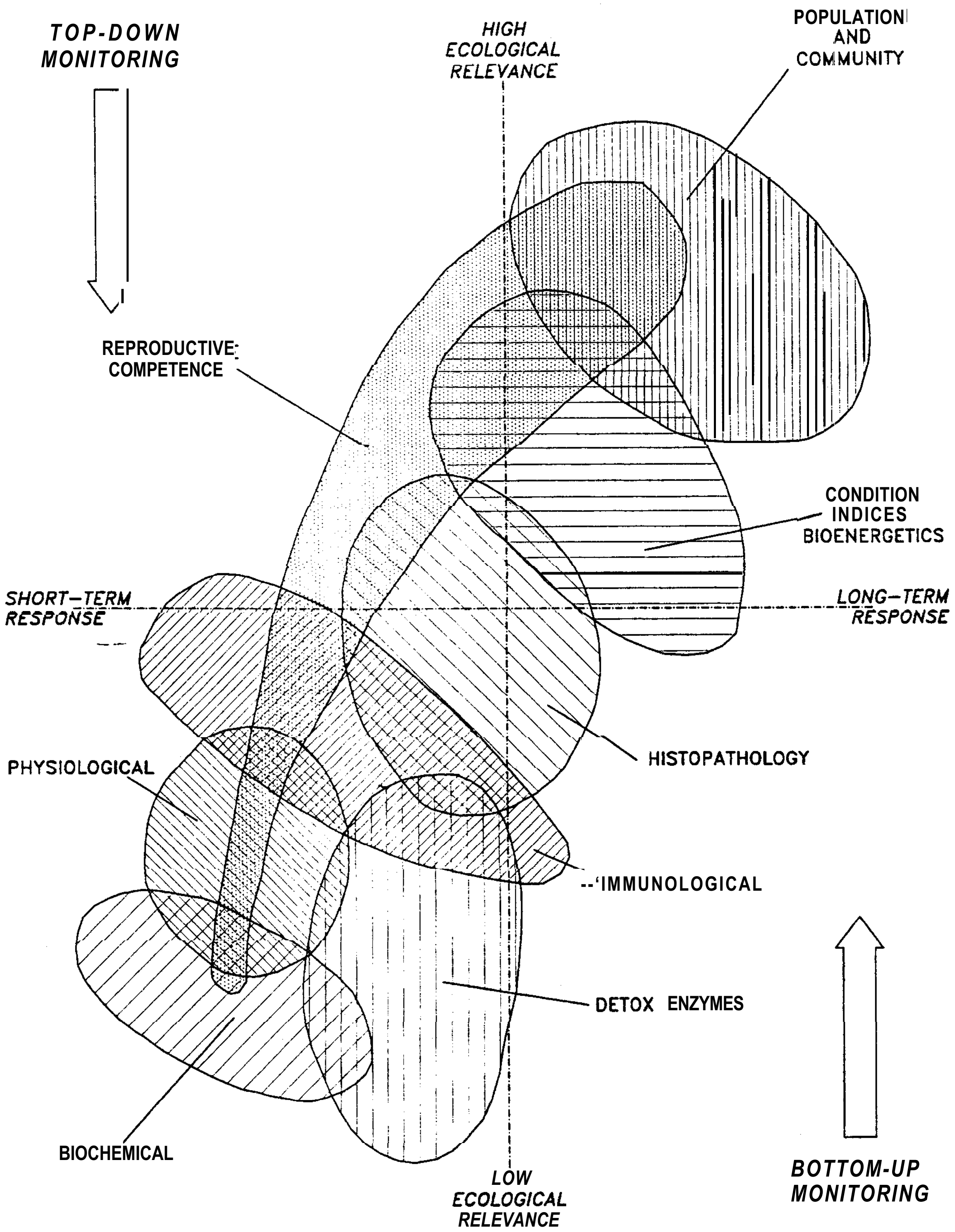
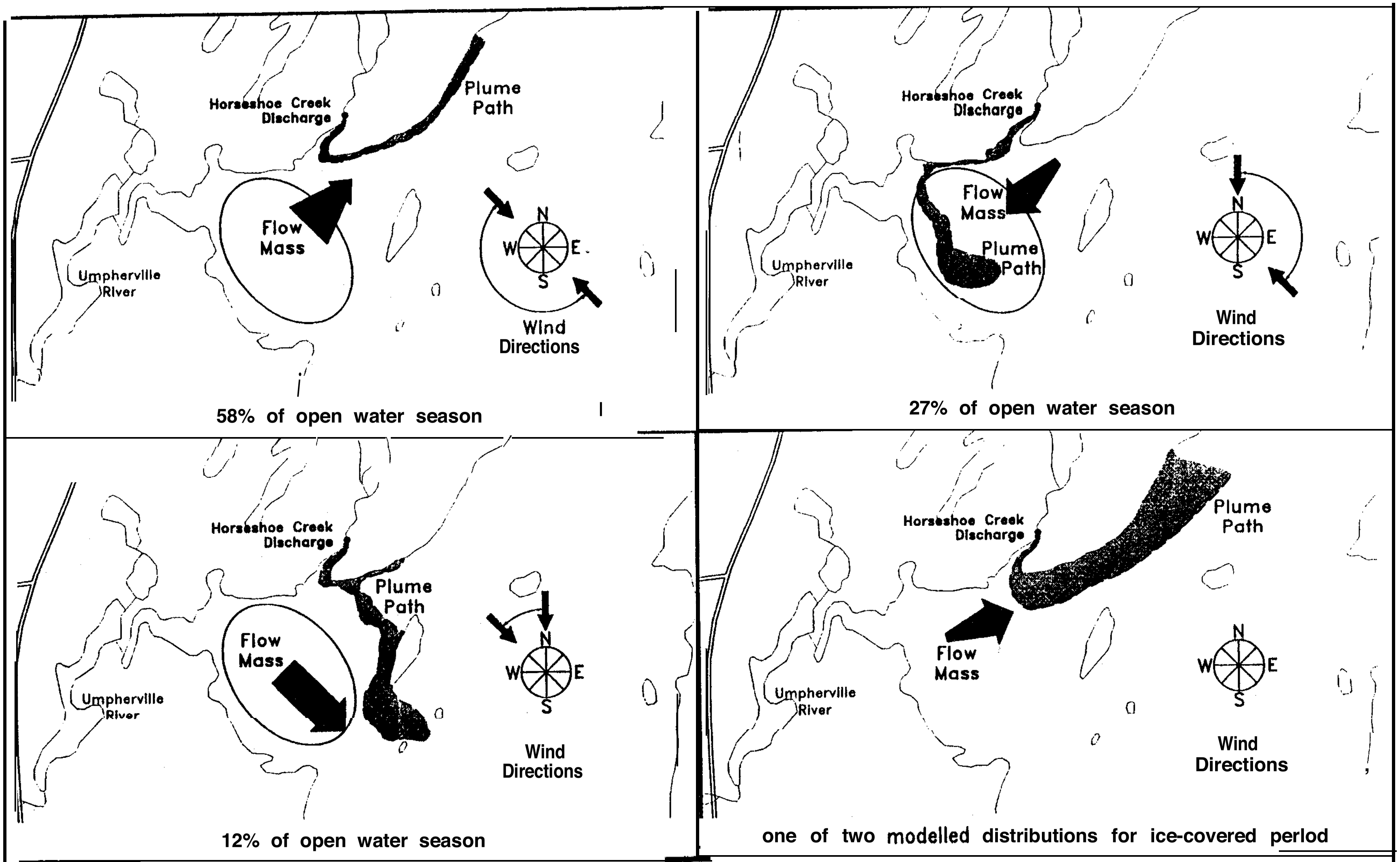


FIGURE IV: 0-1
Organizational levels for monitoring ecosystem health using a top-down or bottom-up approach. Spatial patterns are a conceptual representation of the relationship of the various organizational levels along gradients of response time and ecological relevance (Adams 1990).



58% of open water season

27% of open water season

12% of open water season

one of two modelled distributions for ice-covered period

**FIGURE IV: .1-**
**Modelled** effluent plume distribution patterns exhibiting seasonal variations due to prevailing wind conditions and seasonal discharge patterns (modified from Golder 1994).

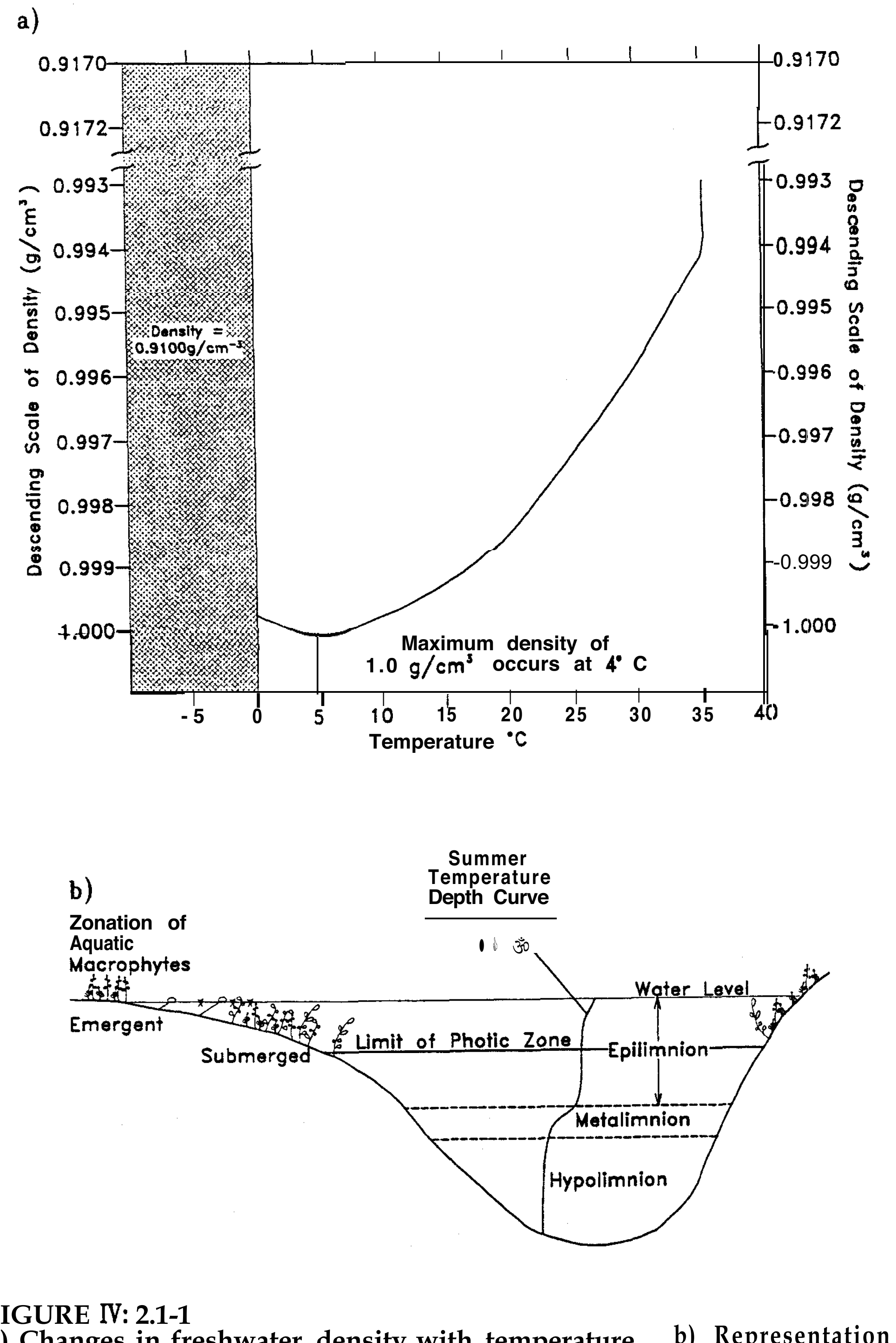


FIGURE IV: 2.1-1
a) Changes in freshwater density with temperature. b) Representation of a lake showing summer stratifications, vegetation zones and the photic zone.

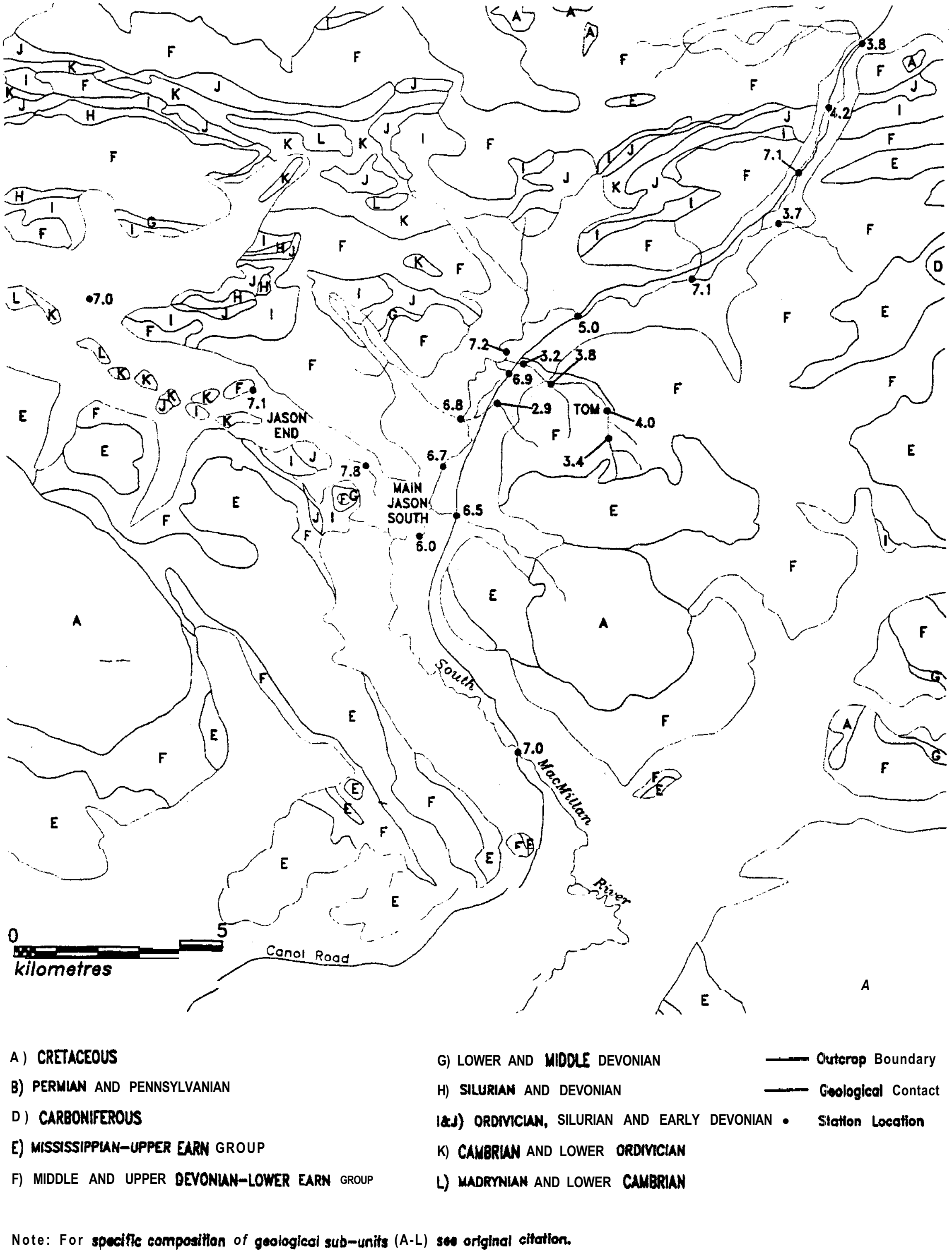


FIGURE IV: 2.1-2
Correlation of water pH to surface geology from the Macmillan Pass region of the Yukon. Note: high acidity in the stream associated with the Tom property receives AMD from exploration activity. (MEND 1.32.1 1993).

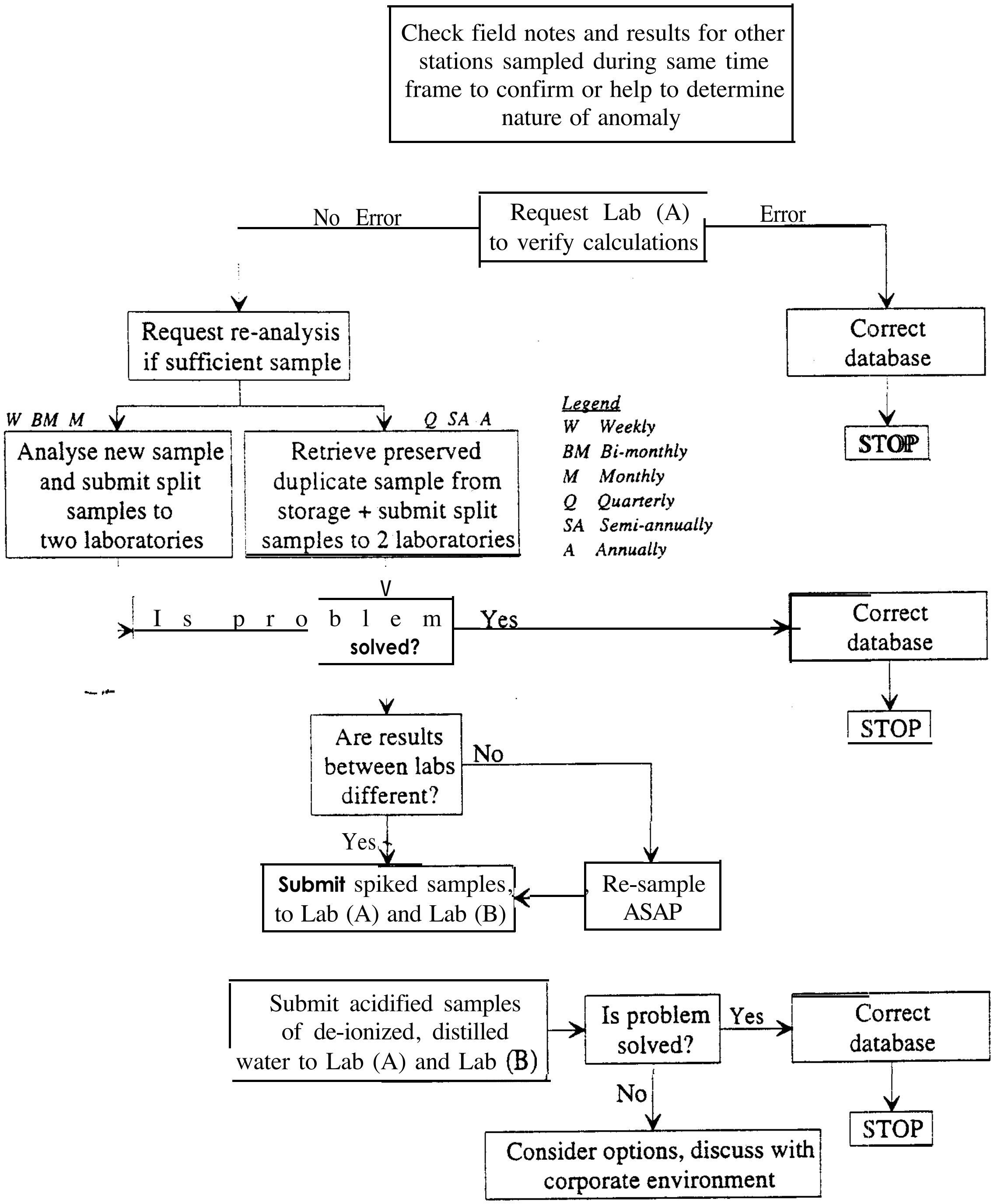


FIGURE IV: 2.1-3
An example of an action flow chart for anomalous water quality results.

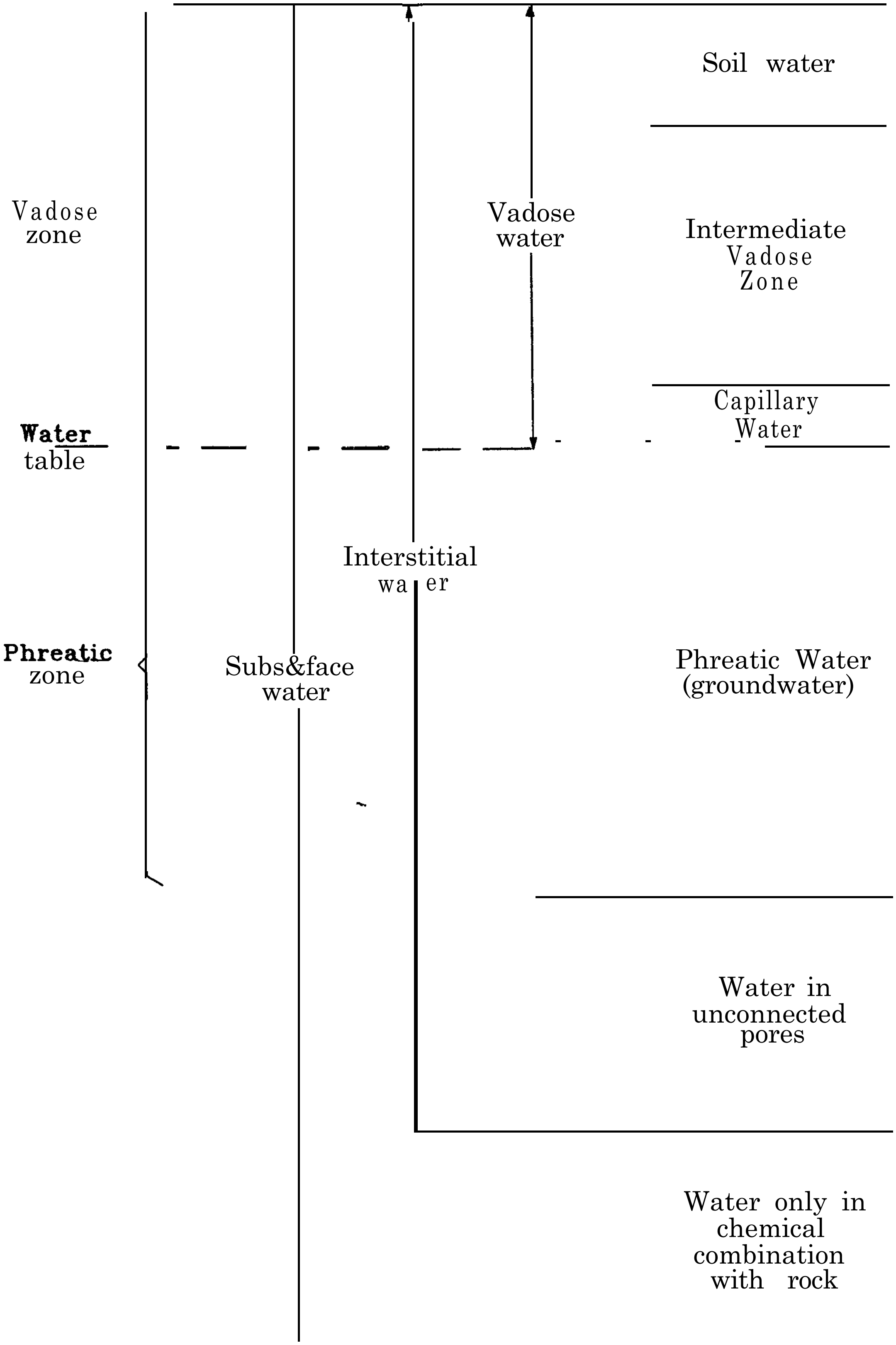


FIGURE IV: 2.2-1
Classification of subsurface water. (Davis and Dewiest in Driscoll, 1989)

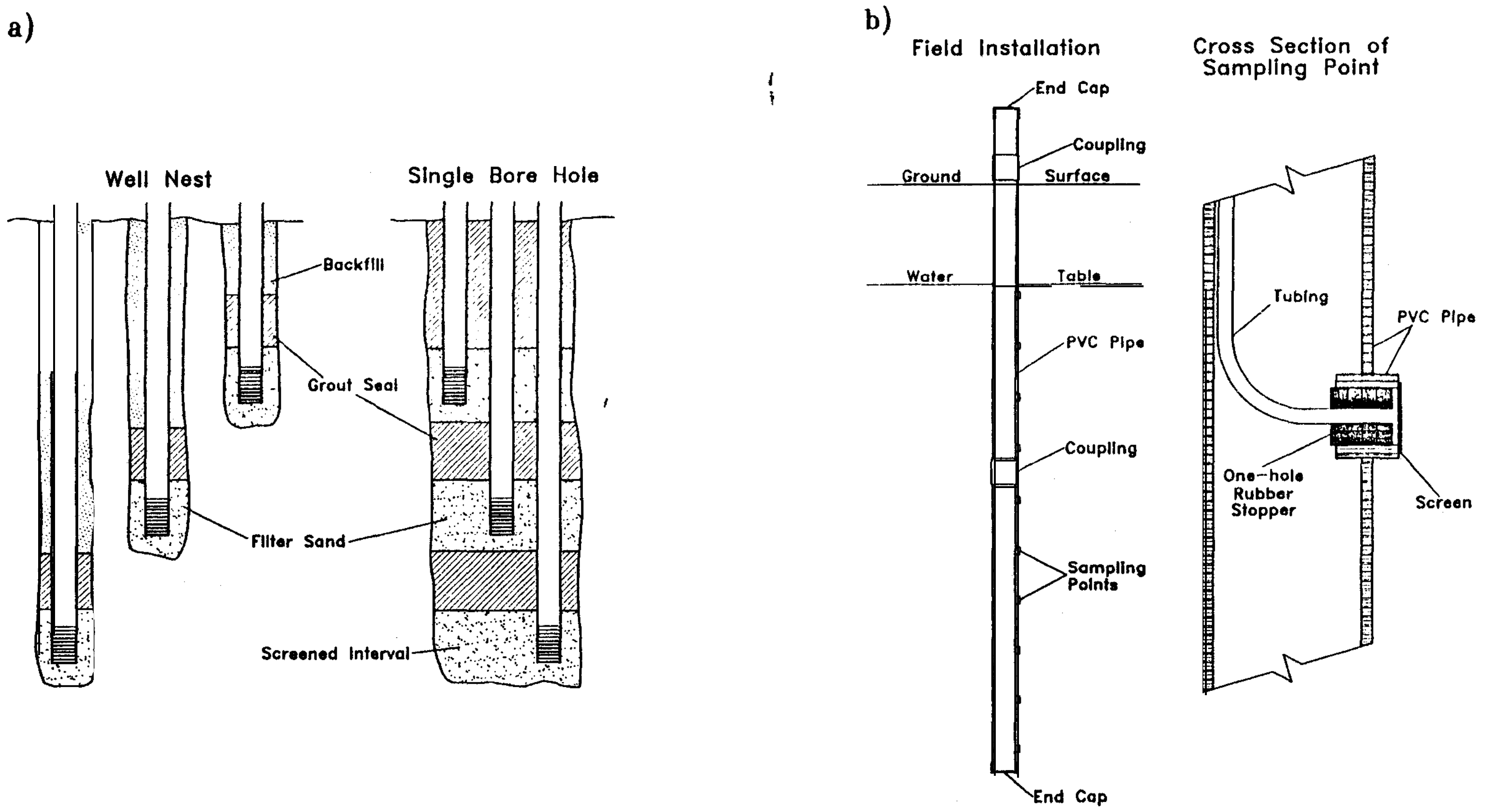
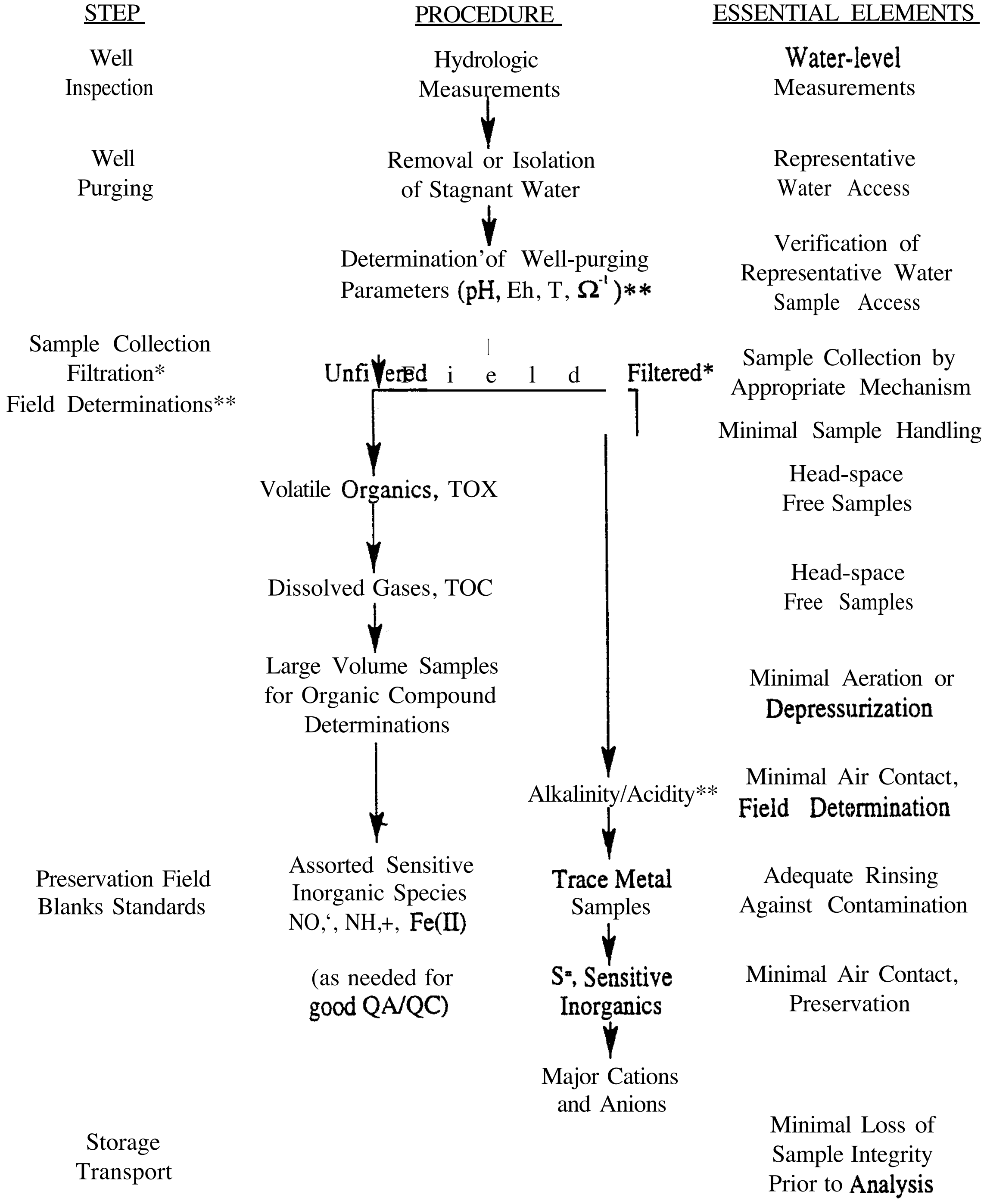


FIGURE IV: 2.2-2
Schematic diagrams of a) standpipe wells in single and multiple installations and b) a multiple port piezometer (US EPA 1991b).



* Denotes samples which should be filtered in order to determine dissolved constituents. Filtration should be accomplished preferably with in-line filters and pump pressure or by $N_2$ pressure methods. Samples for dissolved gases or volatile organics should not be filtered In instances where well development procedures do not allow for turbidity-free samples and may bias analytical results, split samples should be spiked with standards before filtration. Both spiked samples and regular samples should be analyzed to determine recoveries from both types of handling.

**Denotes analytical determinations which should be made in the field

FIGURE IV: 2.2-3 Generalized flow diagram of ground-water sampling steps.

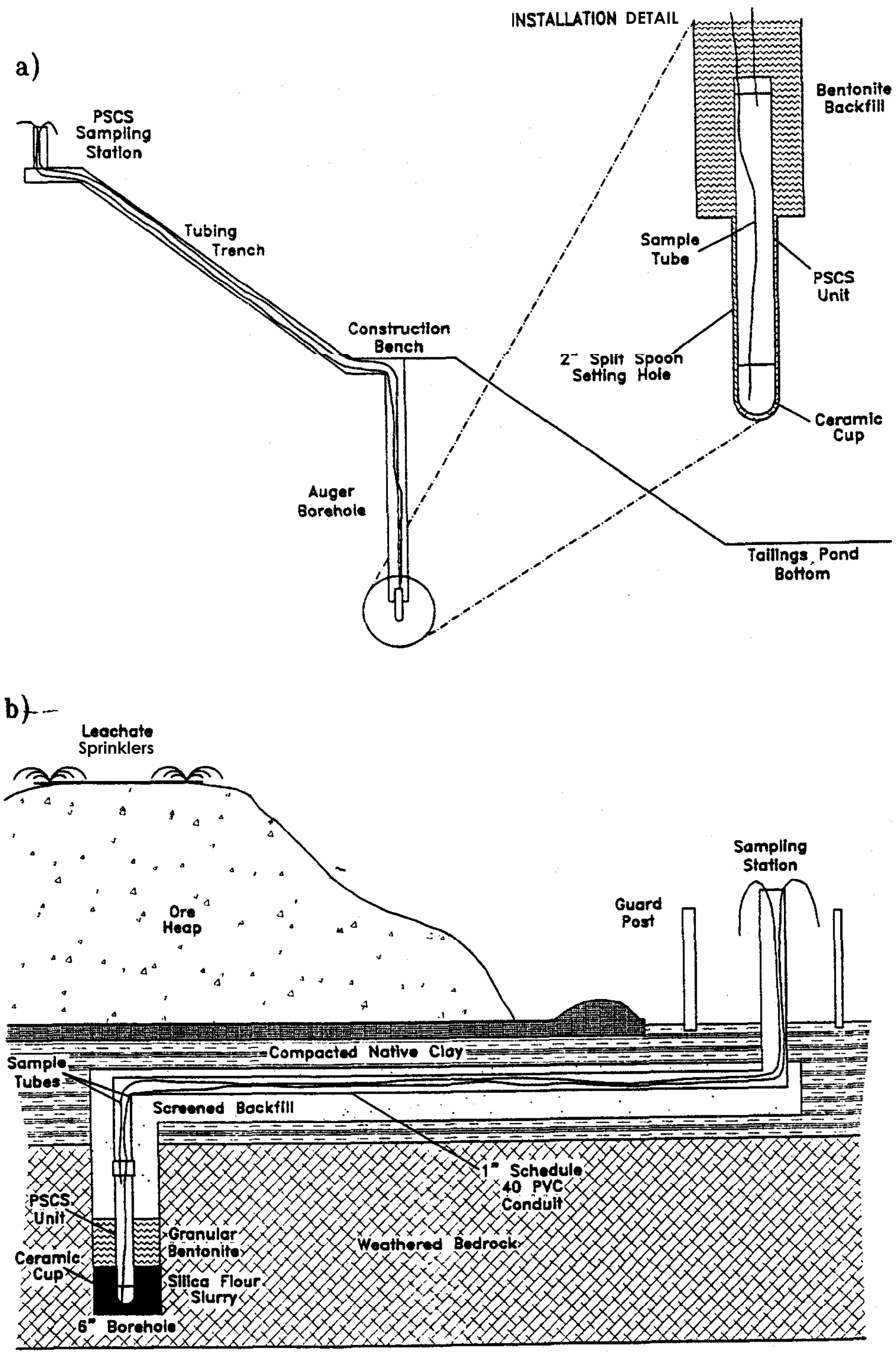


FIGURE IV: 2.2-4
Two examples of vadose zone monitoring a) Installation of porous suction cup samplers (PSCS) around the margins of a uranium tailings pond as an early warning leak detection system. b) Vadose monitoring beneath a leach heap facility (Bond 1995).

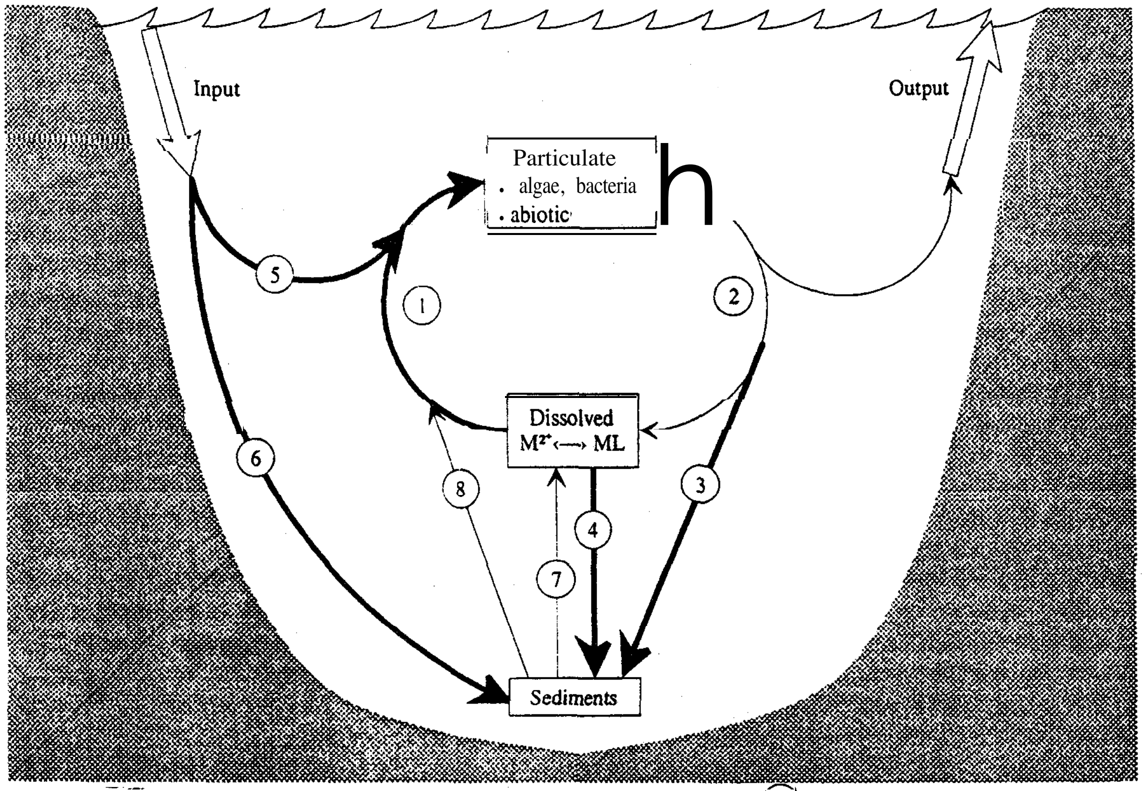


FIGURE IV: 3.1-1a ① see section IV: 3.1 For **further** details
Simplified biogeochemical cycle for trace **metals** in lakes (Hart 1982).

→ Uptake (concentrating) pathways

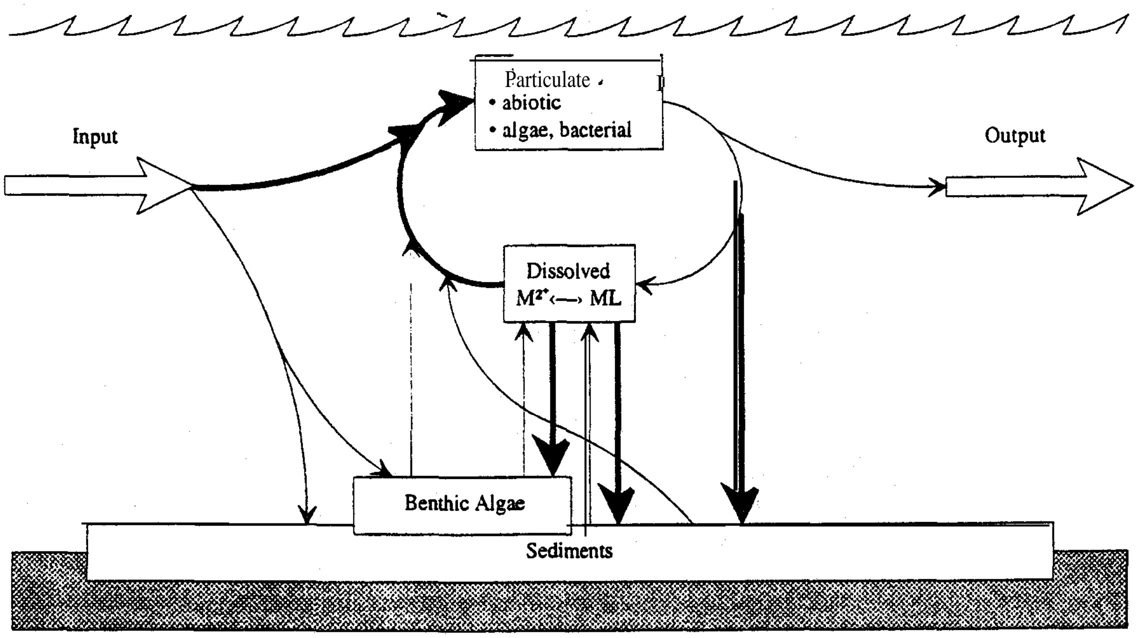


FIGURE IV: 3.1-1b
Simplified biogeochemical cycle for trace metals in **fluvial** systems including the interaction with benthic algae (Hart 1982).

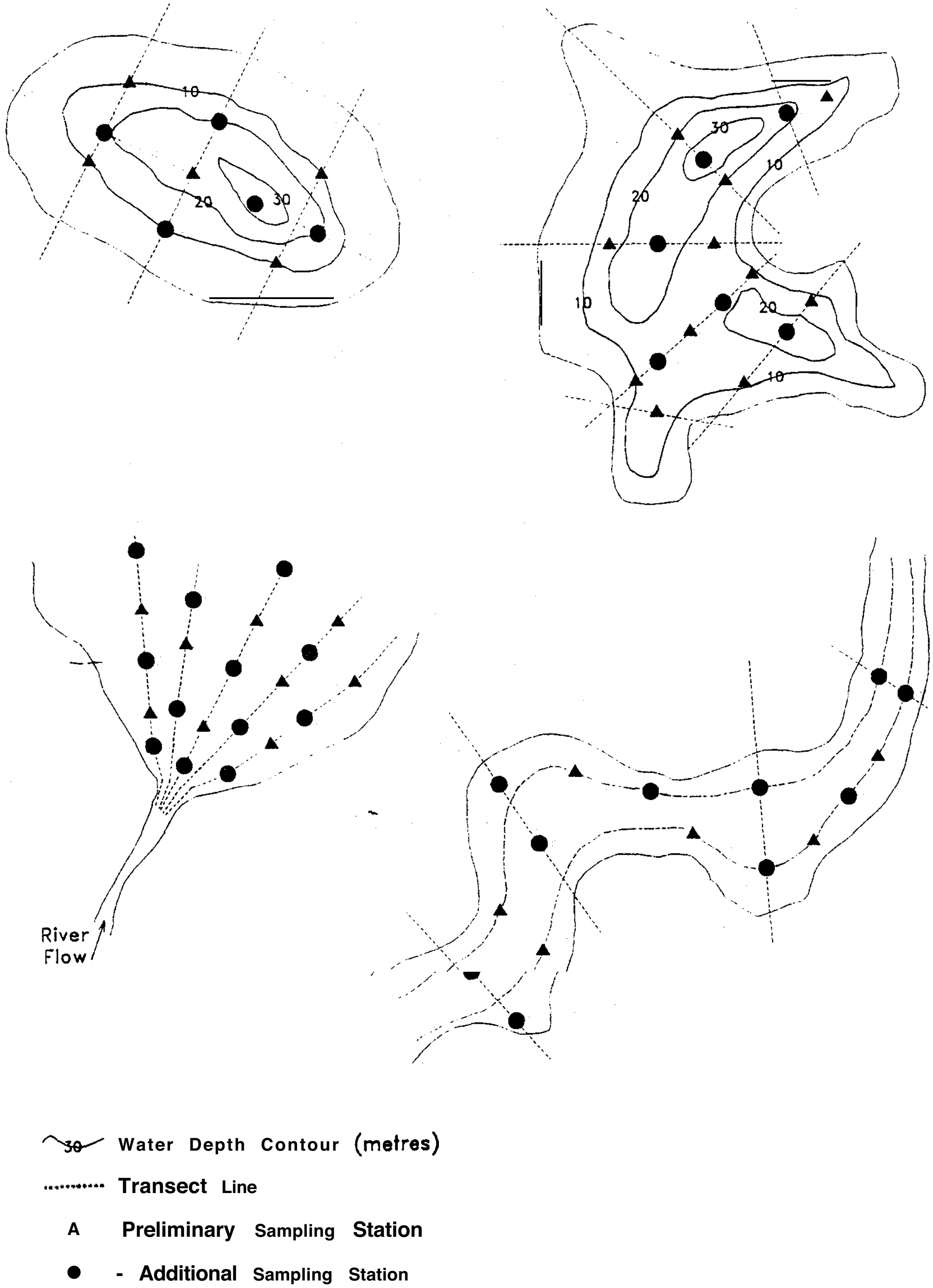


FIGURE IV: 3.4-1
Design of preliminary sediment sampling in different waterbodies (Mudroch and Azcue 1995).

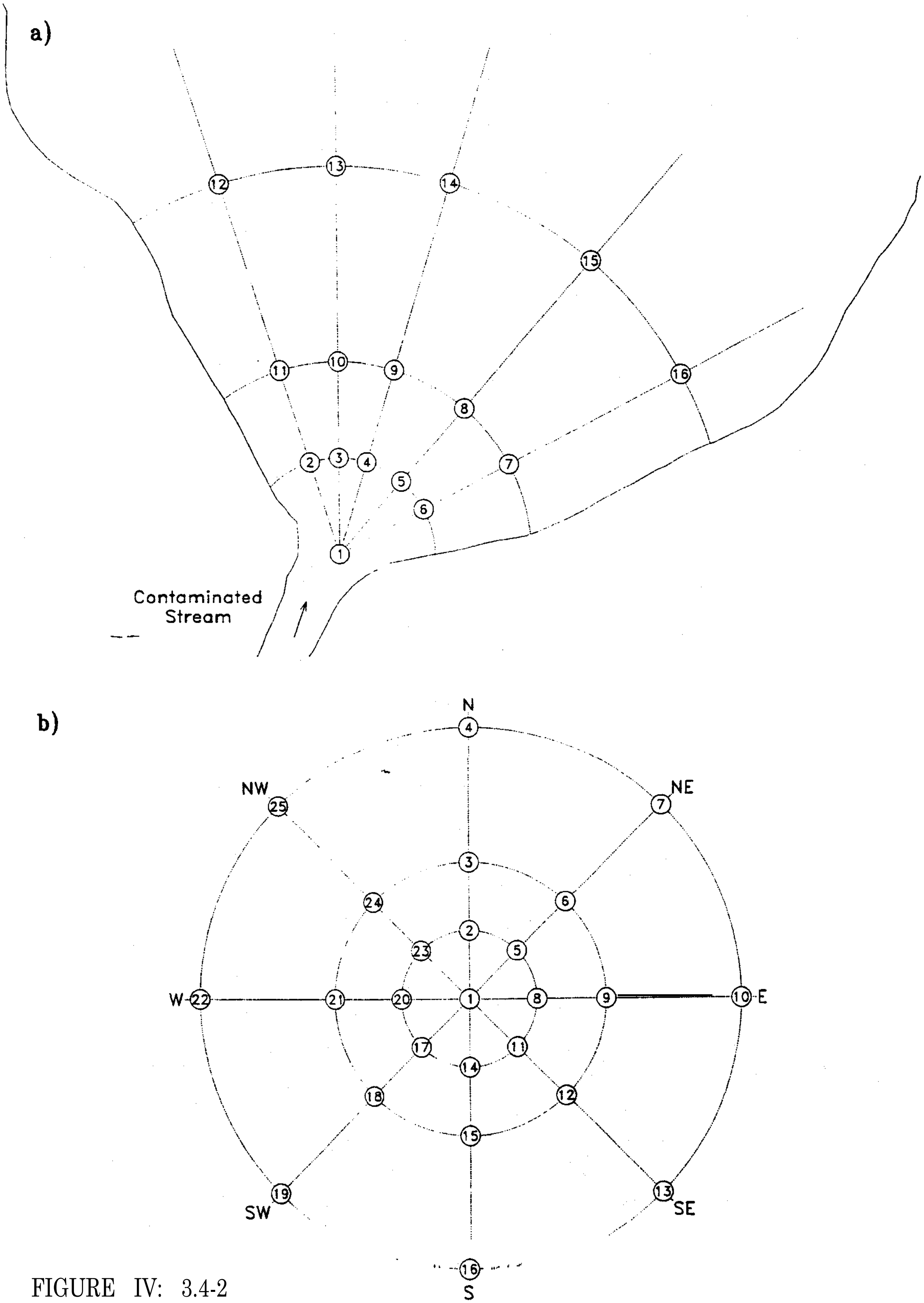


FIGURE IV: 3.4-2
Location of sediment stations for **monitoring** source contaminants, a) contaminant release into a large lake, b) contaminants deposited into the profundal zone of a waterbody.

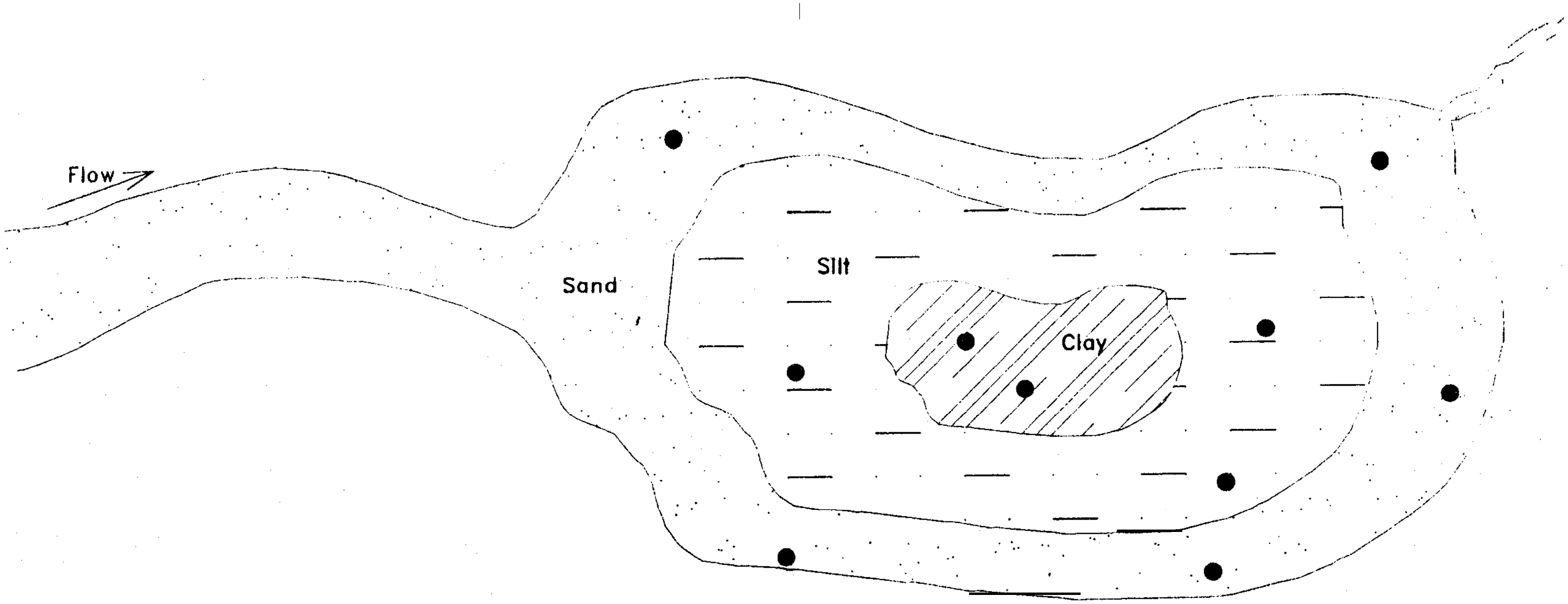


FIGURE IV: 3.4-3
Study-site divided into strata based on certain selected characteristics (e.g., particle size). Stations then randomly located within the strata. Number of stations placed within a strata may be, 1) based on the spatial contribution of the strata to the study area as a whole or, 2) the number required to estimate a mean selected contaminant value within a specified confidence interval for each stratum.

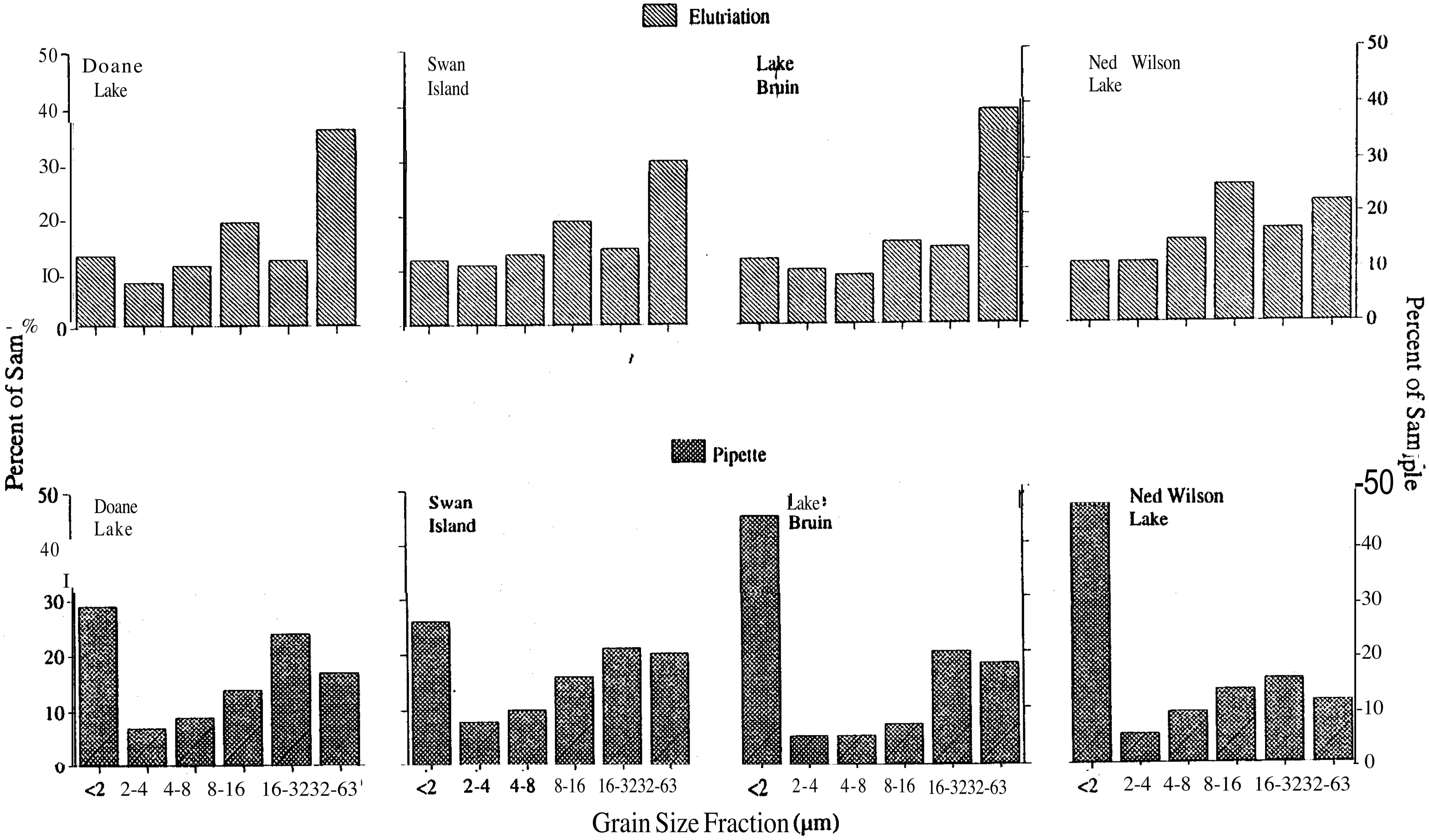
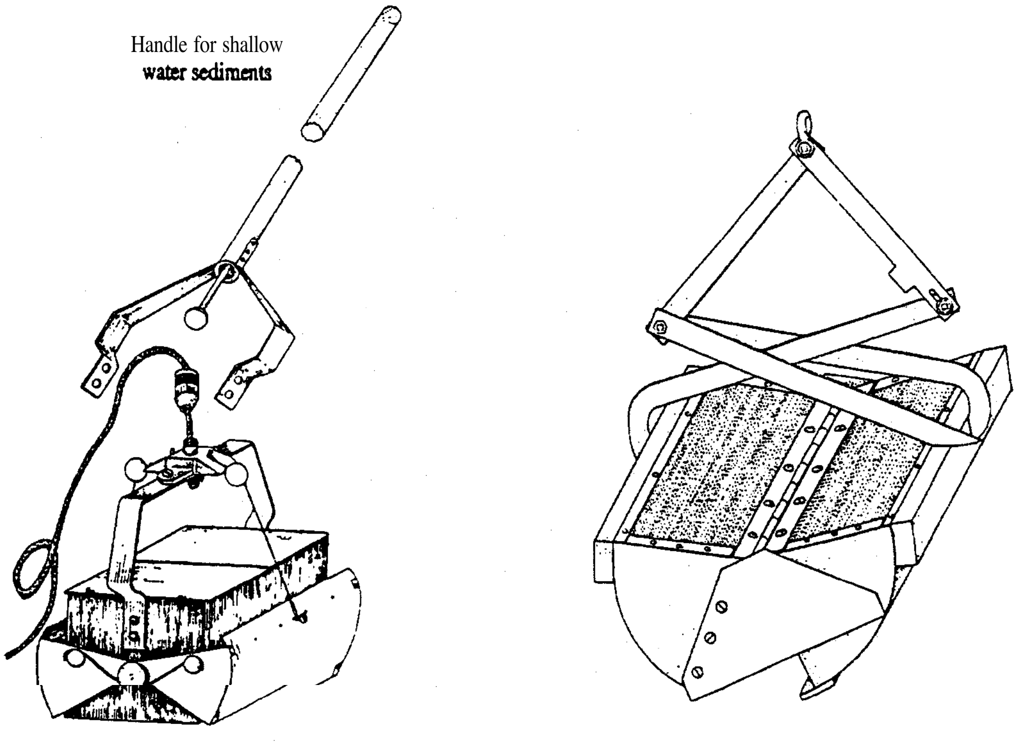
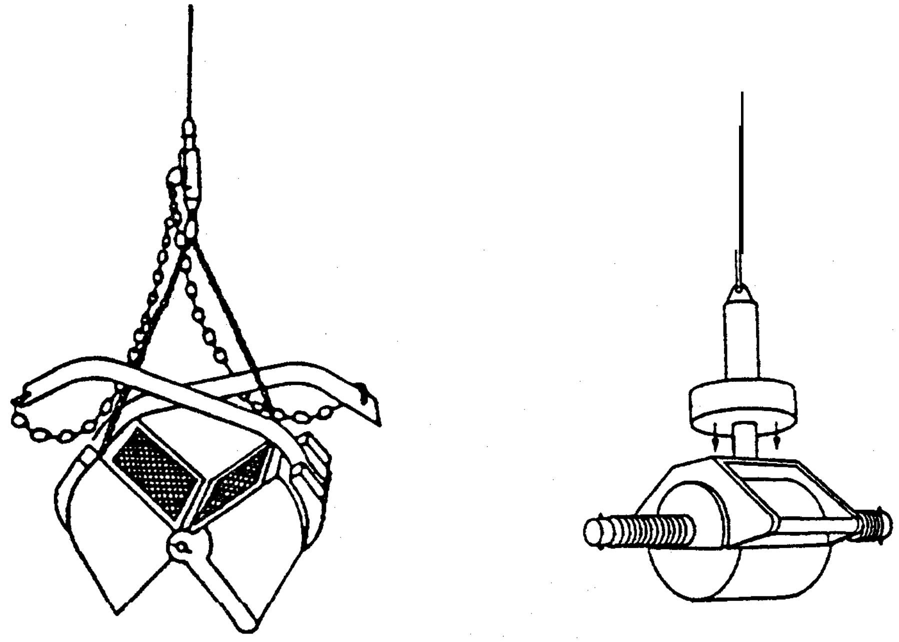


**FIGURE** IV: 3.4-4
An example of the sensitivity of particle size distribution determinations to the specific seperating technique. Note the differences in particle size distributions resulting when air elutriation and dispersed pipette procedures are compared (Horowitz and Elrick 1986).



**Ekman Grab Sampler**

**Ponar Grab Sampler**



**Van Veen Grab Sampler**

**Shipek Grab Sampler**

FIGURE IV:3.4-5
Examples of commercially available grab samplers (diagrams from Merritt and Cummins 1984 and Mudroch and Azcue 1995).

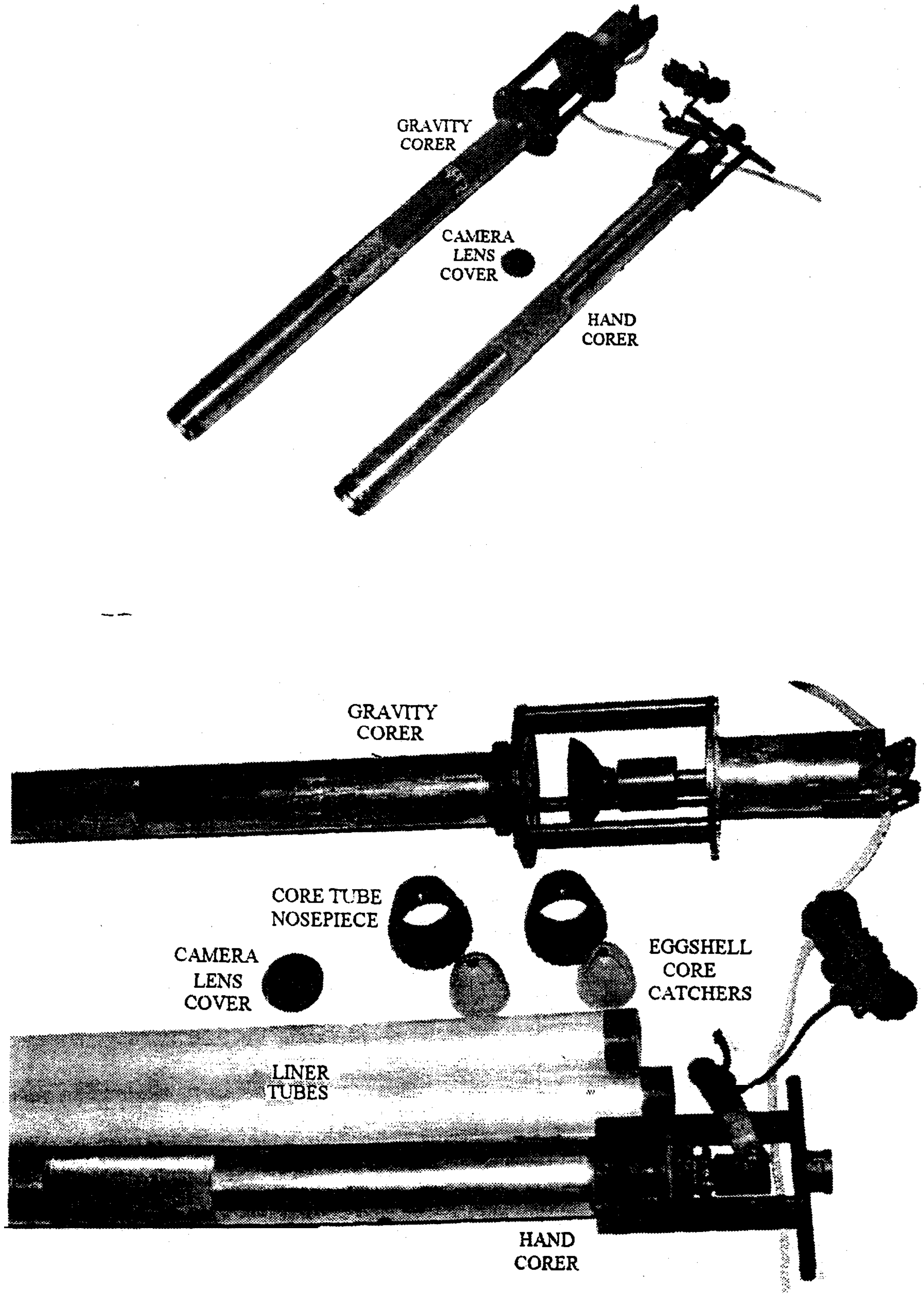


FIGURE IV: 3.4-6 Gravity and hand corers with accessories (TAEM 1996).

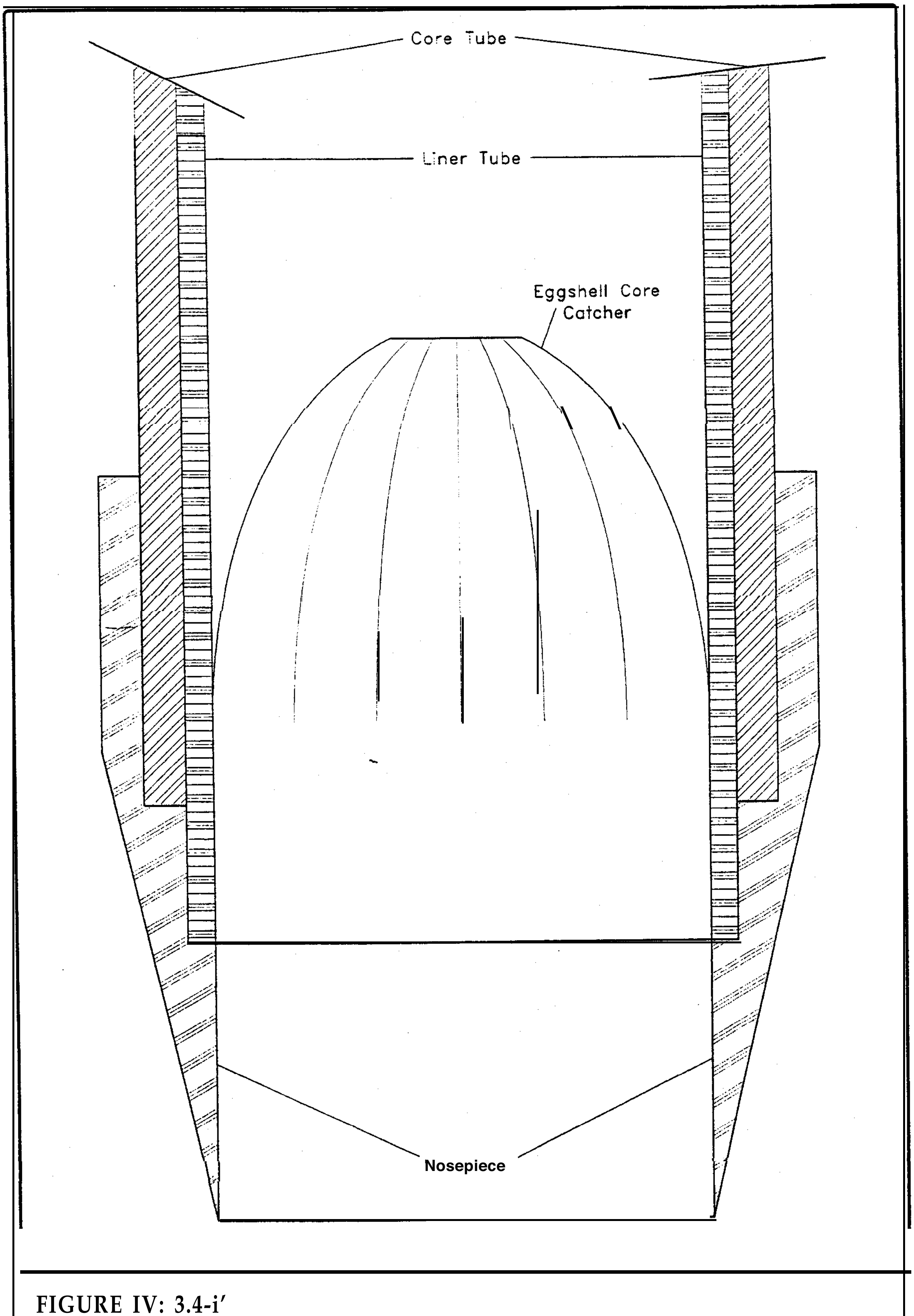


FIGURE IV: 3.4-i′
Cross-sectional view of sediment sampler lower end (TAEM 1996).

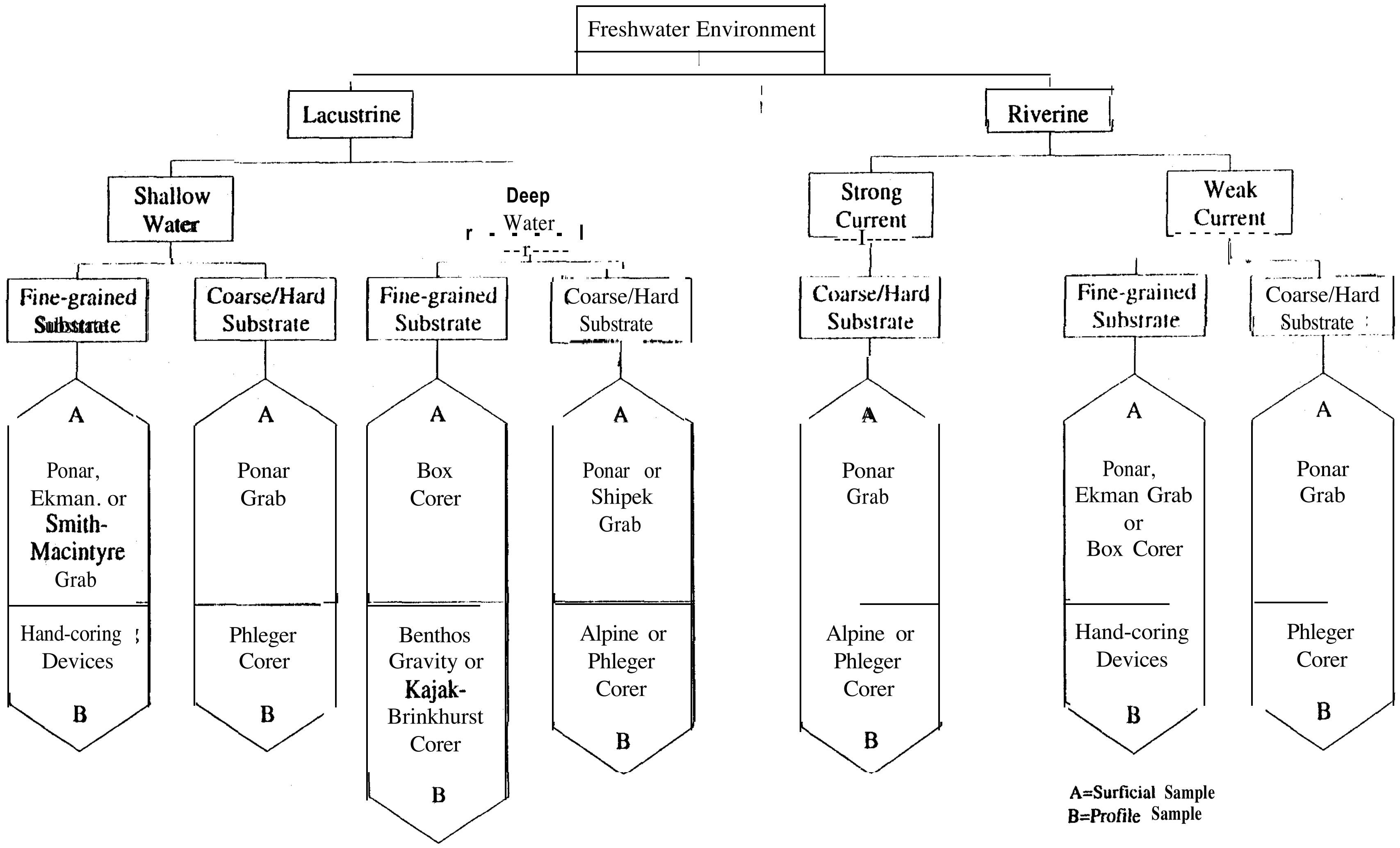
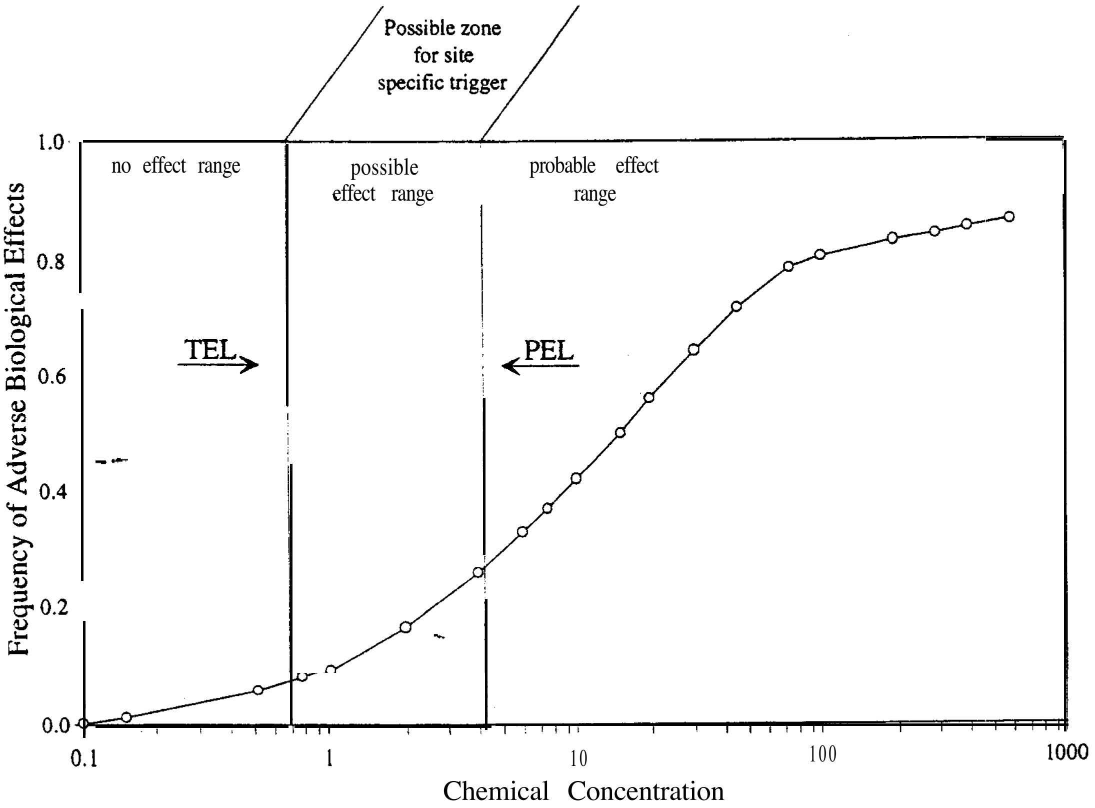


FIGURE IV: 3.4-8 Recommended samplers for different types of freshwater envjronments.



TEL = Threshold Effects Level
PEL = Probable Effects Level

FIGURE IV: 3.4-9 Conceptual example of effect ranges for a sediment-associated chemical (Smith et al. 1996).

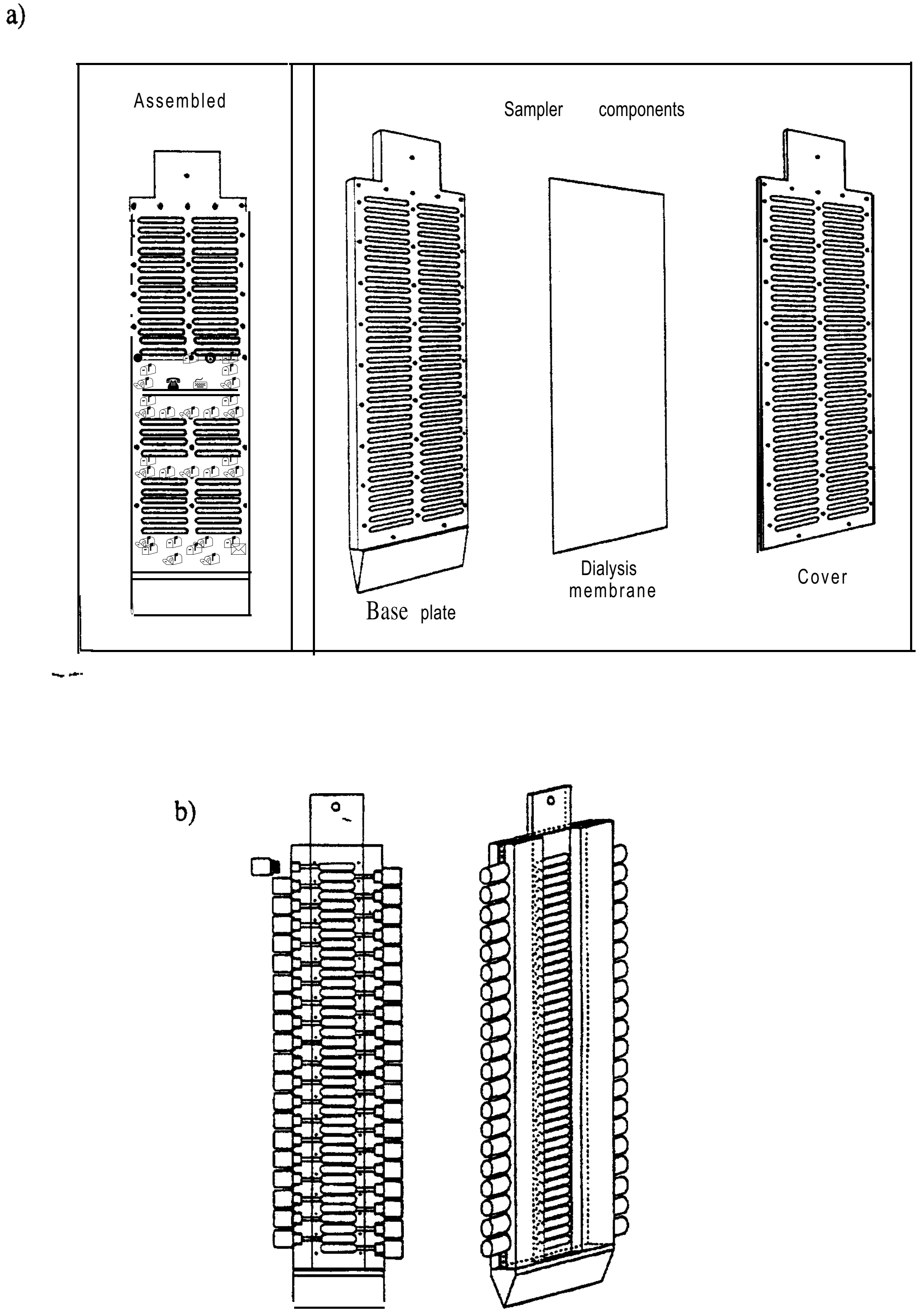


FIGURE IV:3.6-1 Front view and components of a) dialyzer ("peeper") sampler (Mudroch and Azcue 1995) and b) a modified peeper referred to as a Volume Enhanced Sediment Porewater Sampler (VESPOS) (Azcue et al 1996).

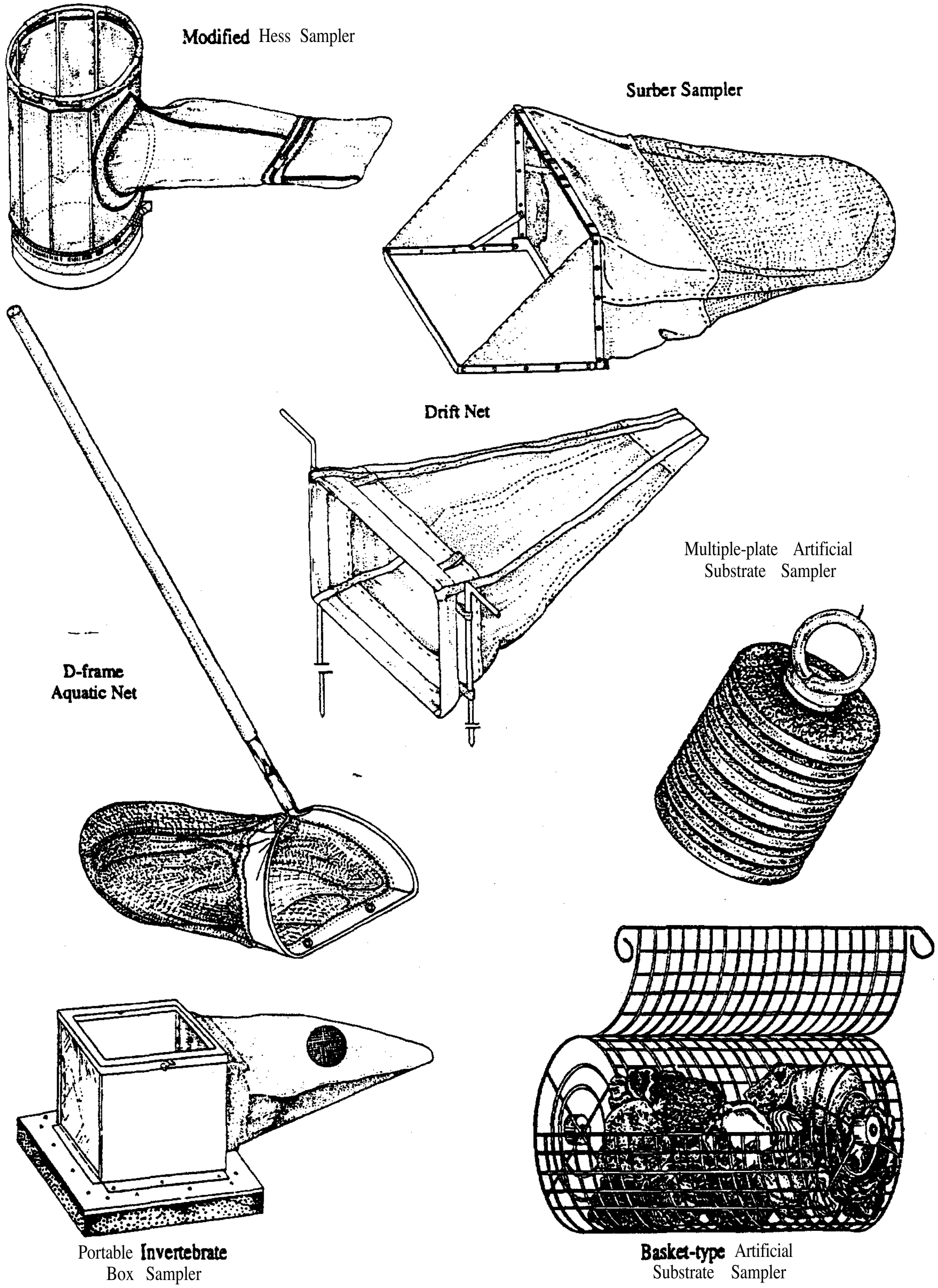


FIGURE IV:4.2-1
Examples of the most common erosional and artificial substrate benthic macroinvertebrate samplers (drawings from Merritt and Cummins 1984).

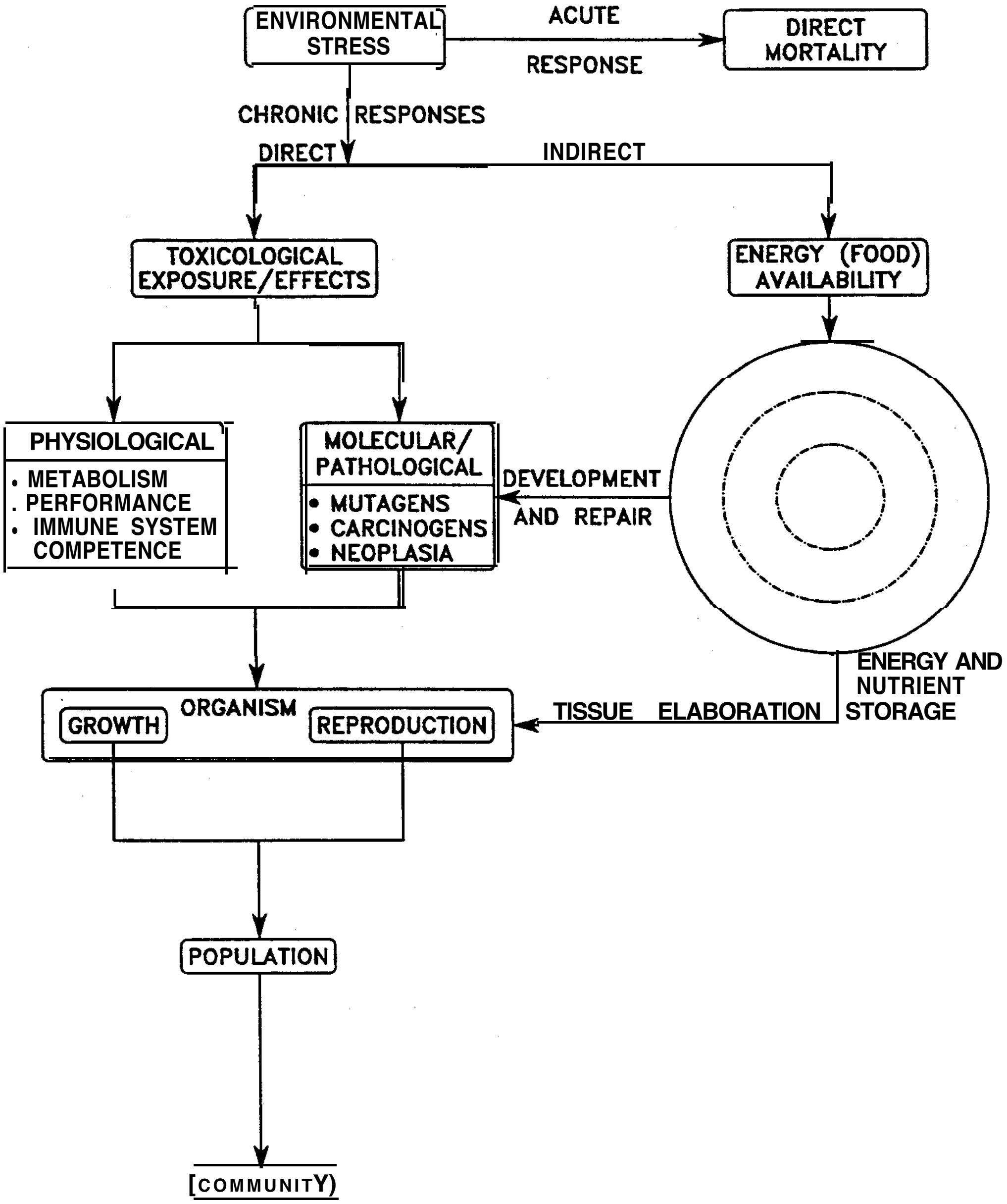


FIGURE IV: 5.2-1
Responses of fish to chronic environmental stress. Emphasis is given to the influences of direct and indirect pathways at suborganism levels on responses expressed at the organism, population, and community levels. The varying diameter of the food compartment signifies the natural seasonal and annual fluctuations in food availability (after Adams 1990).