DESIGN GUIDE FOR THE SUBAQUEOUS DISPOSAL OF REACTIVE TAILINGS IN CONSTRUCTED IMPOUNDMENTS

MEND Project 2.11.9

This work was done on behalf of MEND and sponsored by
Placer Dome Inc.

March 1998

Design Guide

for the Subaqueous Disposal of Reactive Tailings in Constructed Impoundments

Table of Contents

Foreword ix

I - THE GEOCHEMISTRY OF ACID ROCK DRAINAGE AND TRACE METALS 6

1.0 INTRODUCTION 6

2.0 SULPHIDE REACTIVITY 6

- 2.1. Chemistry of Sulphide Oxidation 6
  - 2.1.1. Reactions Defining Pyrite Oxidation 6
  - 2.1.2. Pyrite Oxidation in Subaerial *vs*. Subaqueous Environments 9

3.0 TRACE METAL GEOCHEMISTRY 10

- 3.1. Organic Matter and Sedimentation Rate 10
- 3.2. Geochemical Reactions in the Tailings 10
- 3.3. Chemical Manifestations of Dissolved Metals in Pore-waters 12
- 3.4. Geochemistry of Specific Trace Elements and Mill Reagents 12
  - 3.4.1. Nickel, Copper, Zinc, Cadmium and Lead 12
  - 3.4.2. Arsenic, Antimony and Molybdenum 14
  - 3.4.3. Mercury 16
  - 3.4.4. Cyanide 17

II - DATA COLLECTION AND DATABASE DEVELOPMENT 18

1.0 INTRODUCTION 18

2.0 DATA COLLECTION PROGRAMS 18

- 2.1. Pre-operations Phase 19
- 2.2. Operations Phase 19

III - CHEMICAL CONSIDERATIONS FOR DESIGNING A SUBAQUEOUS TAILINGS IMPOUNDMENT 21

1.0 INTRODUCTION 21

2.0 MINERALOGY 23

3.0 MILLING AND METALLURGICAL PROCESSES 25

- 3.1. Gravity Separation 25
- 3.2. Flotation 25
- 3.3. Pressure Oxidation 26
  - 3.3.1. Neutralization Solids 26
  - 3.3.2. Barren Residues 26
- 3.4. Cyanide Leach Tailings 27

4.0 CHEMICAL TREATMENT OF TAILINGS PONDS INPUTS 29

- 4.1. Treatment of ARD 29

5.0 OTHER INPUTS TO TAILINGS PONDS 30

5.1. Contaminant Mass Balance 31

IV - PHYSICAL CONSIDERATIONS FOR TAILINGS POND DESIGN 32

1.0 INTRODUCTION 32

2.0 WATER COVER DESIGN OBJECTIVE 32

3.0 PHYSICAL PROCESSES AFFECTING BED STABILITY 32

3.1. Processes with Minimal Impact 34

3.2. Wave Action 35

3.3. Ice Entrainment 36

3.4. Sediment Properties 38

4.0 MINIMUM DEPTH CRITERIA 39

4.1. Wave Activity 39

4.2. Ice Entrainment 41

5.0 DESIGN PROCEDURE OUTLINE 41

5.1. STEP 1: Collect Existing Field Data 42

5.2. STEP 2: Collect Topographic Data 44

5.3. STEP 3: Review Technical Advances 44

5.4. STEP 4: Determine Regulatory Requirements 44

5.5. STEP 5: Design and Initiate Data Collection Programs 44

5.6. STEP 6: Obtain Data From Milling/Processing 45

5.7. STEP 7: Assume Initial Pond Dimensions (Width and Length) 45

5.8. STEP 8: Characterize Design Event/Design Return Period 47

5.9. STEP 9: Calculate Minimum Depth Requirement For Bed Stability 47

5.10. STEP 10: Calculate Total Impoundment Volume 48

5.11. STEP 11: Determine Dyke Height Requirement 48

5.12. STEP 12: Estimate Capital Operating Costs 49

5.13. STEP 13: Vary Live Storage Volume/Design Event 51

5.14. STEP 14: Revise Area Sizing/Dimensions of Pond 51

5.15. STEP 15: Calculate Discharge Rate/Finalize Design 51

5.16. Step 16: Operation and Closure Plans 51

6.0 DATA COLLECTION PROGRAMS 54

6.1. Water Cover Requirements 54

6.2. Water Balance Requirements 55

7.0 SITE SELECTION AND DAM DESIGN 55

7.1. Site Selection 56

7.2. Impoundment Layout and Dam Design 56

7.2.1. Upstream Construction 56

7.2.2. Centreline Construction 57

7.2.3. Downstream Construction 57
7.2.4. Filling Curves 57
7.3. Mitigative Designs for Flooded Impoundments 60
7.3.1. Seepage Control and Drain Design 60
8.0 DEPOSITIONAL STRATEGIES 62
9.0 RISK ASSESSMENT 62
9.1. Goals and Objectives 62
9.2. Risk Assessment Definitions and Basic Concepts 63
9.2.1. Risk Assessment Justification 63
9.2.2. Risk Assessment Process 63
9.3. Risk Assessment Analytical Methods 67
9.3.1. Available Analytical Methods 67
9.4. Hazard or Failure Mode Identification 68
9.4.1. Elemental Failure Modes 68
9.4.2. Compound failure modes 68
9.4.3. Flooded Impoundments 69
V - OPERATIONAL MONITORING OF TAILINGS POND 73
1.0 INTRODUCTION 73
2.0 TAILINGS CHARACTERISTICS 73
2.1. ARD Verification 73
2.2. Tailings Solids Characterization 73
2.3. Supernatant 74
3.0 OTHER FLOWS TO POND 75
3.1. Seepage from Waste Rock 76
3.2. Minesite Runoff 77
3.3. Natural Runoff 77
3.4. Mine/Pit Water 78
3.5. Treated Effluents 78
4.0 WATER /SOLID BALANCE IN POND 78
4.1. Water Cover Requirements 78
4.2. Water Balance 79
5.0 POND CHARACTERISTICS 80
5.1. Water Quality 80
5.2. Sediment/Pore-water Geochemistry 80
5.3. Groundwater 81
5.3.1. Physical Hydrogeology 81
5.3.2. Chemical Hydrogeology 81

6.0 POND DISCHARGES 82
7.0 SUMMARY 82
8.0 GEOTECHNICAL MONITORING 82
VI - CLOSURE MANAGEMENT AND OTHER CLOSURE OPTIONS 85
1.0 INTRODUCTION 85
2.0 PRE-CLOSURE MANAGEMENT 85
3.0 POST-CLOSURE MANAGEMENT 85
  3.1. Water Balance 85
  3.2. Other Flows 85
  3.3. Pond Water Quality 85
  3.4. Groundwater 86
  3.5. Geotechnical Monitoring 86
4.0 OTHER CLOSURE OPTIONS 86
  4.1. Intact Water Cover 86
  4.2. Removal of Water Cover 86
  4.3. Lake Disposal and Submarine Deposition 86
APPENDICES A1
APPENDIX A - CASE HISTORIES A2
A1.0 SITES STUDIED A2
  A1.1 Mandy Lake, Manitoba A2
  A1.2 Benson and Keogh Lakes, BC A3
  A1.3 Anderson Lake, Manitoba A4
  A1.4 Buttle Lake, BC A5
  A1.5 Equity Silver Tailings Pond, BC A5
  A1.6 The Island Copper Mine, BC A6
  A1.7 The Black Angel Mine, West Greenland A8
A2.0 SITE INTER-COMPARISONS AND INFLUENCE OF ENVIRONMENTAL PARAMETERS A12
  A2.1 Dissolved Oxygen, Organic Matter and Redox Potential A12
  A2.2 Time A13
APPENDIX B - DEFINITION OF PHYSICAL VARIABLES AND MINIMUM DEPTH METHODOLOGY B13
B1.0 PHYSICAL VARIABLES B13
  B1.1 MINIMUM DEPTH CRITERIA B13
    B1.1.1 Measured Variables B13
    B1.1.2 Derived Variables B2
    B1.1.3 Output Variables B2
  B1.2 POND DESIGN B3

B1.2.1 Measured Variables B3
B1.2.2 Derived Variables B3
B1.2.3 Site Design Variables B4
B1.2.4 Output Variables B5
B2.0 DERIVATION OF THE MINIMUM DEPTH METHODOLOGY B6
B2.1 WAVE PARAMETERS B6
B2.2 DEPTH PARAMETER B6
B2.3 THRESHOLD OF MOTION B7
B2.3.1 Komar and Miller (1975a,b) B7
B2.3.2 Dingler (1979) B8
B2.3.3 Empirical Equation Derived from Threshold of Motion Data B8
B2.4 PREDICTION OF THE MINIMUM DEPTH B9
B2.5 SETTLING VELOCITY AND RESIDENCE TIMES B9
APPENDIX C - CONTRASTING BEHAVIOR OF OXYGEN IN AIR AND WATER C1
C1.0 The Contrasting Behavior of Oxygen in Air and Water and the Implication to Sulphide Oxidation C1
C2.0 Laboratory versus Field Studies: Are They Comparable or Compatible? C2
APPENDIX D - DATABASE COMPILATION AND APPLICATION D1
D1.0 INTRODUCTION D1
D2.0 DIRECT AQUEOUS INPUTS D1
D2.1 Tailings Slurry Water D1
D2.2 Waste Rock Drainage D1
D2.3 Mine Water D2
D2.4 Minesite Drainage D2
D2.5 NATURAL SURFACE FLOW D2
D3.0 INDIRECT (SOLID PHASE) INPUTS D2
D4.0 DATABASE APPLICATION D3
APPENDIX E - SCIENTIFIC/TECHNICAL REFERENCES E1
E1.0 MEND RESEARCH REPORTS E1
E2.0 CITATIONS E3
E3.0 ENGINEERING STANDARDS E8

List of Tables

Table I-1 Calculated Benthic Effluxes of Dissolved Metals in Various Natural and Tailings Sediments 15
Table I-2 Calculated Benthic Effluxes of Dissolved Arsenic in Various Natural and Tailings Sediments. 16
Table II-1 Summary of Types of Information Generated from the Chemical and Physical Considerations for Tailings Pond Design and Operating and Closure Plans 20

Table III-1 Secondary Minerals Identified in Sulphide-Rich Tailings 24
Table III-2 Possible Inputs to a Tailings Pond resulting from Various Treatment Methods for Cyanide Destruction and Recovery 28
Table III-3 Possible Inputs to a Tailings Pond Resulting from Various ARD Treatment Methods 30
Table IV-1 Processes Affecting Bed Stability 35
Table IV-2 Key Variables Affecting Minimum Depth Criteria 38
Table IV-3 Key Variables Affecting Water Balance of Pond Design 55
Table IV-4 Failure and Accident Causes For Tailings Dams 70
Table IV-5 Elemental Failure Modes for Impoundment Systems 71
Table IV-6 Compound Failure Modes and their Causative Elemental Failure Modes 72
Table V-1 Typical Physical Parameters in Tailings Supernatant 75
Table V-2 Proposed Sampling Frequency for Various Tailings Pond Inputs and the Variables which may Dictate Changes to Monitoring Strategies 77
Table V-3 Variables Affecting Minimum Depth Criteria 79
Table V-4 Variables Affecting Water Balance for Pond Design 79
Table V-5 Proposed Sampling Frequency for Various Pond Characteristics and the Considerations which may Dictate Changes to Monitoring Strategies. 84
Table A1 Summary of Subaqueous Tailings Deposits Examined in MEND Studies A3
Table A2 Composition of the Black Angel Ore and Tailings in Various Years A9
Table A3 Analyses of Seawater and Process Water in Various Years A10
Table B1 Threshold of Motion Data from Literature B11
Table C1 Concentrations and Enrichment Factors of Oxygen in Air and Water as expressed by different measures concentration C1
Table D1 Predicted Loadings and Resulting Pond Concentrations of Zinc and Copper in a Hypothetical Tailings Pond. D4

List of Figures

Figure I-1 Schematic Model Outlining the Reactions Involved in the Oxidation of Pyrite. 8
Figure I-2 Three Hypothetical Concentration Profiles for Dissolved Metals in Tailings Deposits or Natural Sediments. 13
Figure III-1 Establishing a Final Design for the Subaqueous Disposal of Sulphide-Rich Tailings 22
Figure IV-1 Tailings Pond Cross-Section 33
Figure IV-2 Wave Interaction 37
Figure IV-3 Design Logic Chart. 43
Figure IV-4 Recurrence Interval Risk 46
Figure IV-5 Water Balance 50
Figure IV-6 Operational Logic Chart 52
Figure IV-7 Closure Logic Chart 53
Figure IV-8 Tailings Pond Layouts 58
Figure IV-9 Schematic of Various Methods of Dam Construction 59
Figure IV-10 Pond Filling Curve 61
Figure IV-11 Summary of Risk Assessment Process 65
Figure IV-12 Risk Management Approach Modified 66

Foreword

The Design Guide is directed at a wide range of audiences including regulatory agencies in the developed and developing world, small, intermediate and large mining companies, environmental consultants, non government organizations (NGO's) and the general public. As some of these groups are not highly experienced in the technical issues relating to tailings systems, very broad and general coverage has been included in areas such as the geotechnical aspects, hydraulic design and risk assessment. This information is included in order to illustrate how the detailed geochemical design information can be integrated to yield a complete design package.

The logic flow charts described are intended as a guide to those seeking to make a preliminary evaluation of the applicability of the subaqueous disposal system to their particular project. These charts can also be used to identify variables which can be optimized in establishing the basic configuration of the impoundment. After this prescreening, the project proponent may then retain specialized design consultants to assist in the development of the final detailed design.

As a further guide, references are provided to documents that contain information on detailed design procedures and standards for such factors as risk assessments and geotechnical and hydraulic designs which are described in general terms in this Guide. Readers are encouraged to consult the references to obtain a more complete understanding of the complexities of impoundment design and construction. Where needed, appropriate consulting expertise should be retained to develop a proper and complete tailings system design.

A DESIGN GUIDE FOR THE SUBAQUEOUS DISPOSAL OF REACTIVE TAILINGS IN CONSTRUCTED IMPOUNDMENTS

INTRODUCTION

The practice of disposing of sulphide-rich mine tailings under water has been based on the assumption that the water cover inhibits the generation of acid and the associated release of metals into natural waters. Primary concern has been focused on the potential deterioration of receiving water quality and the impact on biological resources from excessive concentrations of dissolved metals. Though based on sound theoretical principles and on the well understood geochemical characteristics of natural water bodies and sediments, the assumption was largely untested prior to 1989 and had been supported by only a few, limited scientific studies. Since that time the hypothesis that subaqueous disposal prevents acid generation and metal release has been carefully examined in research undertaken under the auspices of the British Columbia Acid Rock Drainage Task Force and the Mine Environment Neutral Drainage (MEND) program. With very few exceptions this work has shown that subaqueous disposal is a valuable technique in limiting the production of acid rock drainage (ARD).

Based on these research results, this manual has been compiled as a design and operations guide for the subaqueous disposal of sulphide-rich tailings in constructed, permanently-flooded tailings impoundments. The focus is on preventing ARD and inhibiting the potential movement of dissolved metals from tailings pore-waters into waters overlying the tailings. Note, however, that the immobilization of some specific metals may not always be possible. Metals such as arsenic and antimony have a unique and well-documented natural cycle that influences their movement into and out of tailings sediments. Hence, if water from a tailings pond needs to be discharged in order to maintain a water balance, the discharge may require processing in a treatment plant as part of the overall tailings management system.

While subaqueous tailings disposal is an environmentally sound disposal strategy for most reactive sulphide-bearing materials, it does not provide a universal single solution. Some tailings containing sulphides may not have reactivities that warrant the extra effort in constructing permanently-flooded impoundments.

The design guide has six parts and five appendices. The appendices include descriptions of case histories, design parameter definitions, ARD testing methods, example design calculations, and references. Each part is focused primarily on chemical or physical factors that need to be considered in the designs of both physical impoundments and operating and closure plans. Information on the history of the work which led to this design guide together with a discussion of factors related to tailings disposal in natural, productive lakes or marine environments may be found in references from 4th ICARD (4th International Conference on ARD).

SUMMARY OF OBJECTIVES OF EACH PART

GEOCHEMISTRY OF ACID ROCK DRAINAGE AND TRACE METALS

Part I summarizes basic background chemistry of metal sulphide geochemical reactions which occur within tailings materials as revealed by the seven years of MEND studies (1989-1996). This summary explains the relevant chemistry of how and why subaqueous tailings disposal is generally successful in achieving a chemically-sound, stable tailings system. In addition, Part I identifies some metals which will not be stable under typical pond conditions, highlighting the fact that the subaqueous tailings disposal technology outlined in this design guide is not a universal solution for all mines.

DATA COLLECTION AND DATABASE DEVELOPMENT

Part II emphasizes the importance of assembling and organizing information into a comprehensive Integrated Chemical and Physical Database for ongoing use in the design, operations and closure phases of tailings systems. Typical information for the database is listed in a table together with associated chemical or physical factors considered in designing subaqueous tailings systems. The database is a fundamental tool in designing for closure, a key management strategy which is essential for the successful application of subaqueous tailings systems technology.

CHEMICAL CONSIDERATIONS FOR TAILINGS POND DESIGN

Part III describes a full range of chemical design data characterizing tailings mineralogy for input in the Integrated Chemical and Physical Database. The concept of a contaminant mass balance is introduced as a means of assessing each inflow to the pond and for selecting a final strategy which will achieve a chemically stable tailings system. Examples of mass balance calculations are developed in Appendix D.

PHYSICAL CONSIDERATIONS FOR TAILINGS POND DESIGN

Part IV describes a full range of physical design data outlining such factors as water depth criteria for input in the Integrated Chemical and Physical Database. A flowsheet is developed for designers to follow in completing the iterative design process fundamental to achieving final designs. The discussion includes descriptions of the key geotechnical factors and pond construction methods most suitable to subaqueous tailings systems. Each physical parameter is defined in Appendix B together with formulae and the derivation of minimum depth methodology. The concept of applying an overall risk assessment to the final design is discussed together with an overview of the available risk assessment methods.

OPERATIONAL MONITORING OF TAILINGS PONDS

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

CLOSURE MANAGEMENT AND OTHER CLOSURE OPTIONS

Part VI briefly outlines the closure monitoring requirements for permanently-flooded subaqueous tailings ponds. The possibility of developing alternative closure strategies through future research on specific sites is also discussed.

PROJECT HISTORY

The information and data in this manual on the prevention and monitoring of acid rock drainage from sulphide-rich mine tailings were generated from research and field work conducted from 1989 to 1996. The research was supported by the British Columbia Acid Drainage Task Force and the Mine Environment Neutral Drainage (MEND) program. The field work was focused on analyzing case studies of tailings already deposited in freshwater environments. References are also made to separate field studies conducted on tailings deposited in marine environments. The ultimate objective was to determine whether or not the disposal of sulphide-rich tailings in constructed, permanently-flooded impoundments or ponds was an environmentally-sound practice. The data gathered focused specifically on the geochemistry of historically deposited tailings. The purpose was to assess the geochemical stability of tailings deposited under water with respect to acid generation and the dissolution and movement of trace metals into covering waters over both the short and long-terms.

This focus did not allow a detailed examination of other environmental issues concerned with tailings disposal, particularly those concerned with deposition under water in natural lake or marine environments. Such issues include impacts to aquatic life, smothering of the benthos during tailings deposition and uptake of trace metals by flora and fauna. Further site-specific research would have to be undertaken on these issues if natural water bodies are considered for tailing deposition.

MEND

The research and analysis work was prompted by governmental and public concerns over the long-term viability of disposing of mining wastes in bodies of water. Though the practice has been undertaken for some decades, it was performed historically for reasons of general convenience of disposal rather than as a control mechanism for reactive tailings. As a consequence, very little scientific evidence was available to show that the practice was in fact viable over the long-term for preventing or controlling acid generation and the release of trace metals from minerals in the tailings to the overlying waters. To address these specific concerns, a research program was initiated in 1989 to help provide answers to three overall questions:

- Although subaqueous tailings disposal in lakes had been undertaken in the past, was the practice scientifically defendable?
- With adequate scientific knowledge and engineered controls, could the practice become acceptable in low-productivity lakes with few or no macro species such as fish?
- With adequate scientific knowledge, engineered controls and specific constraints, could the practice become acceptable in biologically productive lakes?

Outside the MEND program, research into the subaqueous disposal of reactive tailings in marine settings has historically been conducted. The MEND program avoided research on these environments due to the constraints of time and budget, and the complexity of the biological and physical issues such as tailings containment, influence of ocean currents, and influence on biota populations as a result of habitat changes. Nevertheless, the analyses of the literature on tailings deposited in marine basins indicate that geochemical processes there are very similar to those which result in the effective control of reactive sulphide minerals in freshwater environments.

HISTORICAL SOCIAL AND POLITICAL CONCERNS

Historically, the subaqueous disposal of tailings has been controversial, generating many social and political concerns. These are often the result of the lack of adequate information that explains the practical problems associated with tailings disposal and the geochemical processes which can cause, prevent or control potential environmental impacts. As a consequence, a communications program was initiated and technical information disseminated as it was acquired. As part of the program, two workshops, one in 1992 in Winnipeg and another in 1995 in Vancouver, were organized to present research results to government regulators and NGO representatives. In addition, two scientific peer reviews were conducted at different research stages to critique the procedures and results. These programs were critical in obtaining, both from the public and the scientific community, a qualified general endorsement of the project as it proceeded.

In the workshop presentations, the benefits of the subaqueous disposal of tailings were compared with those of alternative disposal methods using conventional disposal sites for reactive tailings which continue to generate acid and trace metals. These workshops were focused on giving participants a broader appreciation of the practical difficulties in controlling acid generation. As a result, participants acquired a better understanding of the merits of subaqueous disposal practices. Subsequently, they gave their initial, but qualified, endorsement of tailings deposition in subaqueous environments such as low-productivity, small headwater lakes or constructed tailings impoundments.

The concerns related to the potential biological and ecological impacts of tailings disposal in natural lakes and in marine settings have yet to be addressed fully. A discussion of the research undertaken in analyzing these two environments follows.

NATURAL BIOLOGICALLY-PRODUCTIVE LAKE SYSTEMS

Though subaqueous disposal of tailings in biologically productive lakes is possible, such disposal would have to be conducted under significant constraints. As the research data and case study analyses described in this manual will show, the disposal of sulphide-rich tailings in biologically productive lakes will not, over the long-term, generate acid or excessive concentrations of dissolved trace metals in the covering water. However, relatively high concentrations of dissolved substances in the tailings slurry will cause a short-term impact on the lake water quality and aquatic life. The development of control or mitigation strategies to address the potential short- or long-term impacts on the biology or ecology of lake systems have not been included in the scope of work for this project. Hence, requests to regulatory agencies for applications to use natural lakes for the subaqueous disposal of tailings will require comprehensive biological and ecological studies of specific lake sites with associated costs borne by the proponent. Additional compensatory costs for replacement of lost habitat may also be a condition of permits. A description of the type and range of studies required to address these concerns will be included as a separate supplementary document to the final Subaqueous Tailings Design Guide.

Tailings disposal in natural lakes may have some long-term advantages for both mine operators and the general public. The use of natural topographic depressions such as those of lakes results in less dependance on the integrity of man-made structures such as dams for the long-term confinement of tailings. Hence such lake tailings systems may have a lower risk of failure relative to totally constructed systems which will be impacted by natural forces and events such as floods, earthquakes, and erosion over the long-term. In addition, while permitting, mitigation, and operating costs may be higher for lakes than for constructed impoundments, the closure costs may be lower due to a simpler system for monitoring and maintenance.

During mine operations and tailings deposition, lake biology and ecology will be unavoidably disturbed. Nevertheless, case studies have suggested that the environment can recover naturally over the long-term. Actions to accelerate the recovery or even improve the original lake productivity could be made part of post-closure rehabilitation plans.

SUBMARINE DISPOSAL

The geochemical behavior of tailings in marine environments has been shown in case studies to be similar to that of tailings in lakes. Marine geochemists assisted in the development of the methodology used in the MEND research program and case study analyses of marine disposal sites provided additional data illustrating the geochemical processes of subaqueous tailings disposal.

No specific work has been conducted by MEND on biological and ecological impacts of tailings disposal in a marine environments and none is currently planned. Future submarine tailings disposal will require very specific and extensive scientific/engineering work to address such additional issues as the influence of currents on tailings confinement as well as impacts on marine life and populations.

PARTICIPANTS

Participants in the MEND research and authors of this design guide are:

- Placer Dome Inc., Department of Environmental Engineering;
- Lorax Environmental Services Ltd.;
- UBC Oceanography, Department of Earth and Ocean Sciences;
- Hay & Company Consultants Inc.;
- Bruce Geotechnical Services;
- Mehling Environmental Services;
- Hudson Bay Mining and Smelting Co. Ltd.

I - THE GEOCHEMISTRY OF ACID ROCK DRAINAGE AND TRACE METALS

1.0 INTRODUCTION

Acid rock drainage (ARD) is a process which occurs naturally and is important in both the biotic and abiotic cycling of sulphur. The key factor in the production of ARD is the exposure of sulphide mineral surfaces to both air and water. These primary ingredients for the oxidation reactions are in abundance on the earth's surface; thus, the acid-generating oxidation process may occur continuously where sulphide minerals are exposed. Naturally-produced acid drainages can have localized impacts, but are generally attenuated by natural neutralizing agents and downstream dilution. Therefore, acidified drainage produced under such conditions may not have a negative environmental impact. The wastes from mining and milling, however, contain relatively high concentrations of metal sulphide minerals and are stored in relatively small, confined areas such as waste dumps and tailings impoundments or ponds. Hence, ARD generated from sulphide-bearing mining wastes has the potential to contribute significant loadings of acidity and dissolved metals to the environment.

The process of metal sulphide oxidation and ARD generation is best understood in terms of the known chemistry of pyrite oxidation. The reactivity of this common sulphide mineral has been studied extensively for several decades from a variety of perspectives including metallurgy, agriculture, aquatic chemistry and environmental science. The well known oxidation chemistry of pyrite provides the basis for understanding the processes of acid generation, the dissolution of metal sulphides, and the mobilization of metal ions. As these topics are central to an understanding of how to minimize the impacts of ARD, they will be discussed in the following sections.

Part I first outlines the chemistry of pyrite oxidation and describes how the oxidation rate in both subaerial and subaqueous environments is ultimately dependent on the availability of molecular oxygen. This is followed by an outline of the geochemistry of trace metals and a description of the sources of trace metals in tailings impoundments. Part I concludes with a summary of the results of research on the geochemistry of trace metals and chemical additives used in the mining and milling of metal sulphide ores.

2.0 SULPHIDE REACTIVITY

Sulphide minerals are formed in reducing environments where oxygen is absent. On the earth's surface, where molecular oxygen is abundant, sulphide minerals are chemically unstable and prone to oxidation, qualifying them as reactive solids. The pathways by which such oxidation proceeds are discussed below.

2.1. CHEMISTRY OF SULPHIDE OXIDATION

2.1.1. REACTIONS DEFINING PYRITE OXIDATION

The oxidation of pyrite, as for many other metal sulphides, can occur under a wide variety of environmental conditions at rates that depend on several variables. The most critical of these variables are partial pressure of oxygen (pO_2), pH, mineral crystallinity, particle surface area, hydrologic conditions and bacterial activity. The key reactions involved in the oxidation process are outlined schematically in Figure I-1.

Although pyrite is unstable in the presence of both ferric iron (Fe^{3+}) and molecular oxygen, Fe3+

oxidizes pyrite more rapidly than O_2 (Singer and Stumm, 1970). Luther (1987) suggested that the rate of oxidation of pyrite was accelerated by the fact that Fe^{3+} can bind chemically to sulphur in the pyrite lattice facilitating direct electron transfer from S_2^{2-} to Fe^{3+}; in contrast, O_2 cannot bind to the pyrite surface. The dissolved Fe^{2+} released to solution undergoes oxidation to Fe^{3+} (reaction 4, Figure I-1). The regenerated iron is then available for further rapid oxidation of pyrite (reaction 2, Figure 1-1). The relatively slow kinetics of Fe^{2+} oxidation make reaction 4 the rate limiting step in the overall sequence. However, due to the decreased solubility of Fe^{3+} at $pH > 4.5$, molecular oxygen has been suggested to be the primary pyrite oxidant at neutral pH. Ferric iron precipitates as insoluble ferric hydroxide ($Fe(OH)_3$) in the presence of oxygen at neutral pH (reaction 5), and as a result, limits the availability of Fe^{3+}.

The notion that molecular oxygen is the primary pyrite oxidant at neutral pH has been challenged by several investigations which have shown that Fe^{3+} is the preferred electron acceptor (Moses *et al.* 1987; Luther, 1987). These results suggest that Fe^{3+} is an effective oxidant even at the low concentrations predicted to be present at neutral pH. It has been proposed that a steady-state concentration of Fe^{3+} can be maintained by a combination of Fe^{2+} oxidation, dissolution of $Fe(OH)_3$ and consumption of Fe^{3+} by pyrite oxidation.

The oxidation of pyrite in acidic environments proceeds at much higher rates than at neutral conditions. This is because the solubility of $Fe(OH)_3$ is greatly increased at low pH and thus oxyhydroxide precipitation does not limit the availability of free Fe^{3+}. Ferric iron therefore, is considered to be the principal pyrite oxidant at low pH, with the rate of pyrite oxidation being solely limited by the regeneration of Fe^{3+} through oxidation of Fe^{2+} by O_2. Since the solubility of metal sulphides also increases with decreasing pH, dissolution processes become more important in acidic environments. The solubility of pyrite, for example, is greatly increased below pH 4.

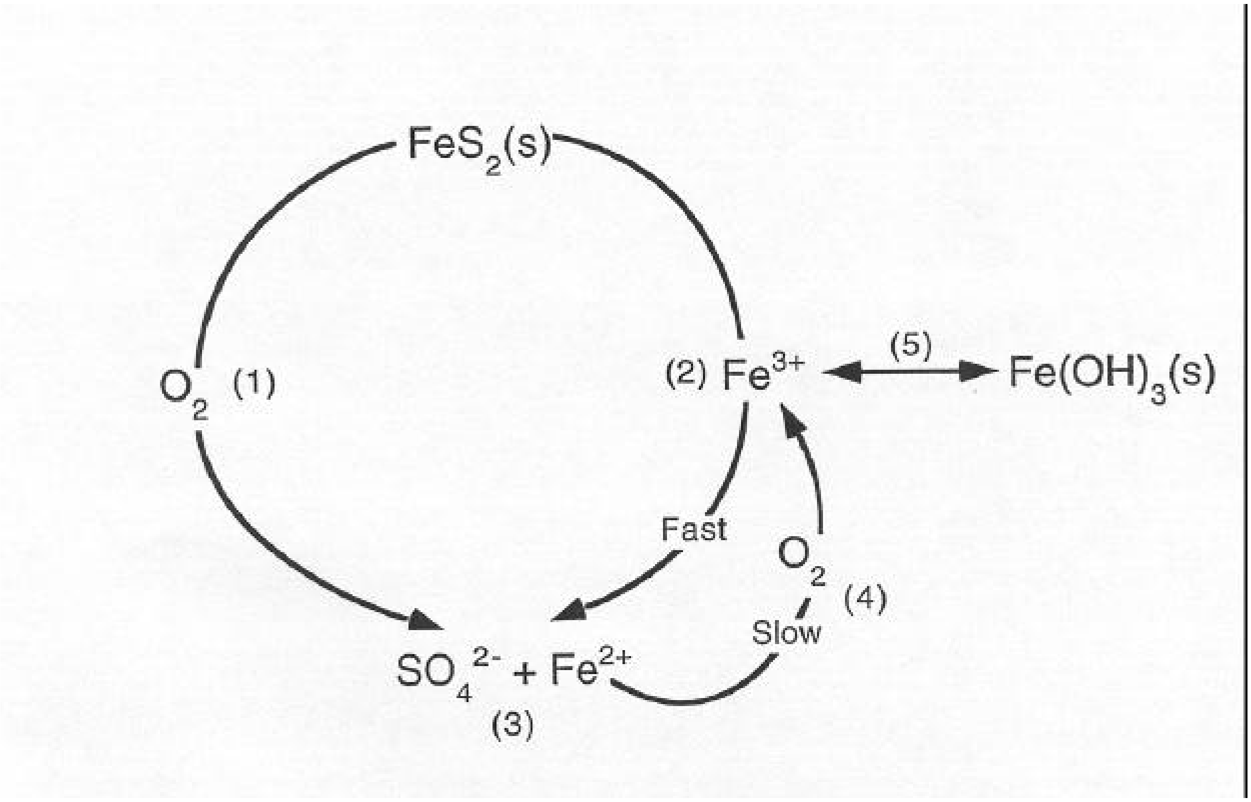


FIGURE I-1.
SCHEMATIC MODEL OUTLINING THE REACTIONS INVOLVED IN THE OXIDATION OF PYRITE.

The precipitation of iron oxide on the surfaces of pyrite can also indirectly retard oxidation by providing a physical barrier to Fe^{3+} and O_2. This concept has been explained in terms of a "shrinking core" model, in which the precipitation of ferric hydroxide progressively limits the diffusional replacement of oxidants at the pyrite surface (Nicholson *et al.*, 1990).

Although the chemical oxidation of Fe^{2+} is very slow at pH < 4.5, the presence of iron and sulphur oxidizing bacteria can accelerate the oxidation process (Singer and Stumm, 1970). *Thiobacillus ferrooxidans*, for example, has been shown to enhance Fe^{2+} oxidation by O_2 by factors reportedly as high as 10^3 to 10^6 (Nicholson, 1994). *T. ferrooxidans* is an acidophilic bacterium which obtains energy for growth via oxidation of ferrous iron, elemental sulphur (S^o), metal sulphides and other reduced inorganic sulphur compounds (Evangelou and Zhang, 1995). Similarly, *Thiobacillus thiooxidans* is an acidophilic bacterium which derives its energy needs from the oxidation of SE and sulphide; *T. thiooxidans*, however, cannot oxidize Fe^{2+} (Gould *et al.*, 1994). As the term "acidophilic" implies, these bacteria are active in low pH environments (*i.e.*, pH < 4.5); bacterially-mediated oxidation of pyrite at circumneutral-neutral pH has been shown to be minor (Jaynes *et al.*, 1984).

T. ferrooxidans can oxidize pyrite either directly on the pyrite surface producing sulphate, or indirectly via Fe^{2+} oxidation to Fe^{3+}. Consequently, bacterial activity increases the rate of oxidation in the former case by increasing the porosity and surface area of the mineral grains (Mustin *et al.*, 1992). In the latter case, bacteria accelerate the oxidation of Fe^{2+} to Fe^{3+}, and thus regenerate the main pyrite oxidant faster than in their absence (Evangelou and Zhang, 1995). In this manner, these bacteria encourage oxidation by diminishing the rate-limitation represented by reaction 4 in Figure I-1.

2.1.2. PYRITE OXIDATION IN SUBAERIAL *VS.* SUBAQUEOUS ENVIRONMENTS

The kinetics of pyrite oxidation in subaqueous environments are markedly slower than under subaerial conditions. Several factors contribute to the observed difference, all of which ultimately stem from the availability of molecular oxygen. Pyrite in mining wastes exposed to the atmosphere, for example, is initially oxidized by atmospheric oxygen which liberates acidity, sulphate and ferrous iron into solution according to the equation:

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

Following the oxidation of Fe^{2+} to Fe^{3+}, acid is additionally generated via the subsequent hydrolysis of Fe^{3+} to $Fe(OH)_{3(s)}$ via the mechanism:

$$Fe^{3+} + 3\,H_2O \rightarrow Fe(OH\;)_{3(s)} + 3\,H^+$$

The oxidation of Fe^{2+} to Fe^{3+} in the subaerial setting is not limited by the availability of molecular oxygen; consequently the hydrolysis of Fe^{3+} and the generation of acid will progressively lower the pH in the absence of readily available neutralizing minerals. The development of low pH conditions further enhances sulphide oxidation due to increased bacterial activity and increased activity of free Fe^{3+}. The high dissolved metal loads characteristic of ARD can be largely attributed to increased solubility of pyrite and other metal sulphides at low pH.

Rates of sulphide oxidation in subaqueous settings are limited by the much lower availability of dissolved oxygen and the fact that bacterial oxidation of pyrite in such environments is minor. Oxygen limitation stems from two fundamental properties of water. First, the maximum concentration of dissolved oxygen found in natural waters is significantly lower than that found in the atmosphere. Second, and more importantly, once the small inventory of dissolved oxygen in water is consumed, it is typically replaced very slowly by processes of molecular diffusion and small-scale turbulence; diffusion of oxygen is nearly 10,000 times slower than in air. Thus, the oxidation of submerged pyrite cannot generate acid at the rates common to subaerial environments.

3.0 TRACE METAL GEOCHEMISTRY

The exchange of trace metals between submerged tailings and the overlying water can be influenced by a number of factors including the presence of soluble mineral phases or alteration products, the presence or absence of organic matter, the presence or absence of iron and manganese oxides, the rate of tailings accumulation, and the precipitation of *in situ* metal sulphide phases. The results of the field work conducted on old tailings deposits in natural lakes suggest the importance of these influences varies between settings.

3.1. ORGANIC MATTER AND SEDIMENTATION RATE

Mine tailings stored under a water cover are either deposited on top of natural sediments (*i.e.*, in lakes) or they are deposited on artificially-created substrates (tailings impoundments). Natural lake sediments comprise complex mixtures of eroded rocks and soils, organic detritus and other phases resulting from post-depositional processes. Organic matter is degraded into its constituent elements (mostly simple molecules containing the elements carbon, nitrogen and phosphorus, by aerobic bacteria which utilize oxygen in the reaction. These oxidation reactions are most intense at the sediment-water interface due to the greater reactivity of freshly deposited organic matter (Middelburg, 1989). As a result, oxygen typically becomes depleted at shallow sediment depths. In other words, the rate of oxygen consumption in the sediments is greater than the supply rate from diffusional transport from the water above the sediments.

In contrast to natural sediments, mine tailings are typically organic-poor despite the presence of minor quantities of organic mill reagents that can accompany tailings. Although the high sedimentation rates and generally low primary productivity of tailings impoundments imply that accumulations of organic matter in tailings deposits should be minor (*e.g.*, Equity Silver Tailings Pond), tailings themselves consume oxygen, presumably via small-scale oxidation reactions involving sulphide minerals and/or trace organics derived from the milling process. However, even in tailings deposits in which oxygen is rapidly depleted (*i.e.*, implying high oxidation rates), effluxes of dissolved metals from tailings to overlying waters have been shown to be negligible. In such situations of rapid oxygen consumption (*e.g.*, Anderson Lake), the evidence gathered to date suggests that the reactions primarily proceed via the oxidation of labile organics as opposed to the oxidation of sulphide minerals. The high sedimentation rates during the operational phase of tailings deposition also have the effect of capping previously deposited tailings and effectively shielding tailings particles buried deeper than a few centimeters by limiting the transfer of oxidants to deeper horizons.

3.2. GEOCHEMICAL REACTIONS IN THE TAILINGS

The principal consumption mechanisms that remove trace metals from pore-waters include sulphide precipitation and sorption to iron and manganese oxides. Conversely, processes which remobilize metals into solution include the oxidation of organic-metal complexes and the reductive dissolution of oxides. These processes, which apply to both natural sediments and tailings, are briefly described below.

Significant accumulations of organic matter in subaqueous tailings deposits enhance the stability of metal sulphides by fostering conditions in which sulphides form naturally. Once oxygen has been depleted, anaerobic bacteria utilize other oxidants to facilitate organic matter degradation for their energy needs. Below the aerobic zone in sediments, the secondary oxidants are, in the order of preference: nitrate, manganese oxide, iron oxide, and sulphate. Of particular importance is the consumption of sulphate which is naturally present in small amounts in natural waters, and is often abundant in mill process water of the tailings slurry. The by-product of the reduction of natural or introduced sulphate is hydrogen sulphide which rapidly reacts with most dissolved metals (*e.g.*, cadmium, copper, mercury, arsenic, molybdenum, nickel, iron, lead and zinc) to produce highly insoluble, solid metal sulphides. The consumption of metals via sulphide precipitation in near-surface sediments can result in a flux (*i.e.*, transport) of dissolved metals into the surface sediments/tailings from the overlying water, with the tailings acting as a sink for dissolved metals rather than a source. Such removal was observed in tailings deposits in Anderson, Mandy and Benson Lakes. The natural tendency in organic-rich deposits, therefore, is towards the creation of an environment in which sulphides form naturally, thereby providing a chemically stable environment for the storage of sulphide-rich mine tailings.

The second factor is referred to as *oxide blocking or oxide scavenging*. Such scavenging involves the oxides of iron and manganese, two of the secondary oxidants discussed above. In the presence of dissolved oxygen, Fe and Mn oxides exist as solids which strongly adsorb many trace metals. In deeper suboxic horizons, these solid phases are utilized as secondary oxidants in the oxidation of organic matter. Upon their reduction in these reactions, Mn and Fe are released back into solution. The liberated Fe^{2+} and Mn^{2+} diffuse upward toward the sediment-water interface where they reprecipitate in oxic sediment horizons to form Fe^{3+} and Mn^{4+} oxides. This cycle of dissolution at suboxic depths, upward migration, and reprecipitation in oxygenated sediments, can result in significant accumulations of Fe and Mn oxides in surface sediments, as seen in Anderson Lake. Since iron and manganese oxides are efficient in adsorbing and co-precipitating a broad range of metal ions, these oxides provide an effective “oxide blocking” mechanism that inhibits the release of dissolved metals from the pore-water upwards and into the water column (Figure I-2c).

During periods of summer stratification or during periods of ice-cover, the restricted input of dissolved oxygen to the bottom waters can result in the depletion of oxygen at the sediment-water interface. The development of these conditions during such periods can result in the dissolution of iron and manganese oxides present in the surface sediments. This mechanism can result in the remobilization of trace metals associated with these solid phases, and their release to the overlying bottom waters. Reoxygenation of the bottom waters promotes the reprecipitation of iron and manganese oxides. This seasonal cycle of oxide dissolution-reprecipitation appears to be common to productive temperate lakes which form ice every winter (*e.g.*, Anderson Lake).

One final barrier to all metal release from tailings within lake sediments is time. The burial of tailings by natural sediments or more recently deposited tailings occurs progressively. As the dominant transport mechanism of dissolved metals in sediments is diffusion, and because mass transport by diffusion is effective only over short distances (*i.e.*, a few centimetres), accumulation of a relatively thin layer of sediments over an abandoned tailings deposit is sufficient to isolate tailings chemically from the water column (*e.g.*, Benson Lake). In this regard, impacts due to subaqueous tailings disposal from the perspective of tailings oxidation is, at worst, a relatively short-term issue and once deposition has ceased and tailings have been buried by a few centimetres of natural sediment, they can be considered to be chemically secure.

3.3. CHEMICAL MANIFESTATIONS OF DISSOLVED METALS IN PORE-WATERS

Tailings pore-waters are little more than lake or pond water trapped between tailings particles; in the absence of chemical reactions, the composition of pore-waters would be identical to the overlying waters. However, various post-depositional processes influence the mobility of trace metals in sediments and tailings deposits. If tailings are reactive and release dissolved metals (*e.g.*, via oxide dissolution), for example, the most sensitive indicator will be locally elevated concentrations of metals within shallow (*e.g.*, via sulphide precipitation) pore-waters (*e.g.*, Figure I-2a). Conversely, precipitation or consumption of dissolved metals is characterized by concentrations that decrease with depth (*e.g.*, Figure I-2b). A difference in dissolved metal concentration across a depth interval defines a concentration gradient, which results in the net migration of dissolved metals from the zone of high concentration to the zone of low concentration. This process occurs through the random motion associated with all dissolved ions or molecules and is termed molecular diffusion. The amount of dissolved metals that migrate along a concentration gradient is termed the flux and is proportional to the concentration gradient.

In order for a dissolved metal to diffuse between the sediments and overlying waters, a concentration gradient must exist across the sediment-water interface. Depending on the slope of that gradient, a metal can be said to be diffusing out of, or into, the sediments. A higher concentration of a metal in the pore-water relative to the overlying water cover indicates an “efflux” of dissolved metal out of the sediments (Figure I-2a). Conversely, lower dissolved metal concentrations in pore-waters relative than benthic waters imply an “influx” of metal into the sediments (Figure I-2b). Equal concentrations of a particular metal on both sides of the interface represent “diffusive equilibrium”, and indicate a zero net transport (Figure I-2c).

3.4. GEOCHEMISTRY OF SPECIFIC TRACE ELEMENTS AND MILL REAGENTS

3.4.1. NICKEL, COPPER, ZINC, CADMIUM AND LEAD

Nickel, copper, zinc, cadmium and lead are elements commonly associated with sulphide ores and exhibit similar geochemical behaviours in subaqueous systems. In oxygenated waters at neutral pH, all five elements exist as divalent cations, and depending on the environment, are associated with various inorganic and organic complexes. Inorganic complexes typical to freshwaters include free hydrated aquo ions (*e.g.*, Ni^{2+}, Zn^{2+}, Cd^{2+}), carbonato species (*e.g.*, $PbCO_3$, $CuCO_3$, $NiCO_3$) and hydroxy complexes (*e.g.*, $CdOH^+$, $CuOH^+$); in seawater, the speciation of dissolved copper and cadmium can be dominated by chloride complexation (Stumm and Morgan, 1981). Associations with organic ligands (*e.g.*, phytoplankton metabolites, proteins, humic acids) have also been shown to be important for this suite of trace metals, particularly for copper, zinc and cadmium (Bruland, 1989, 1992).

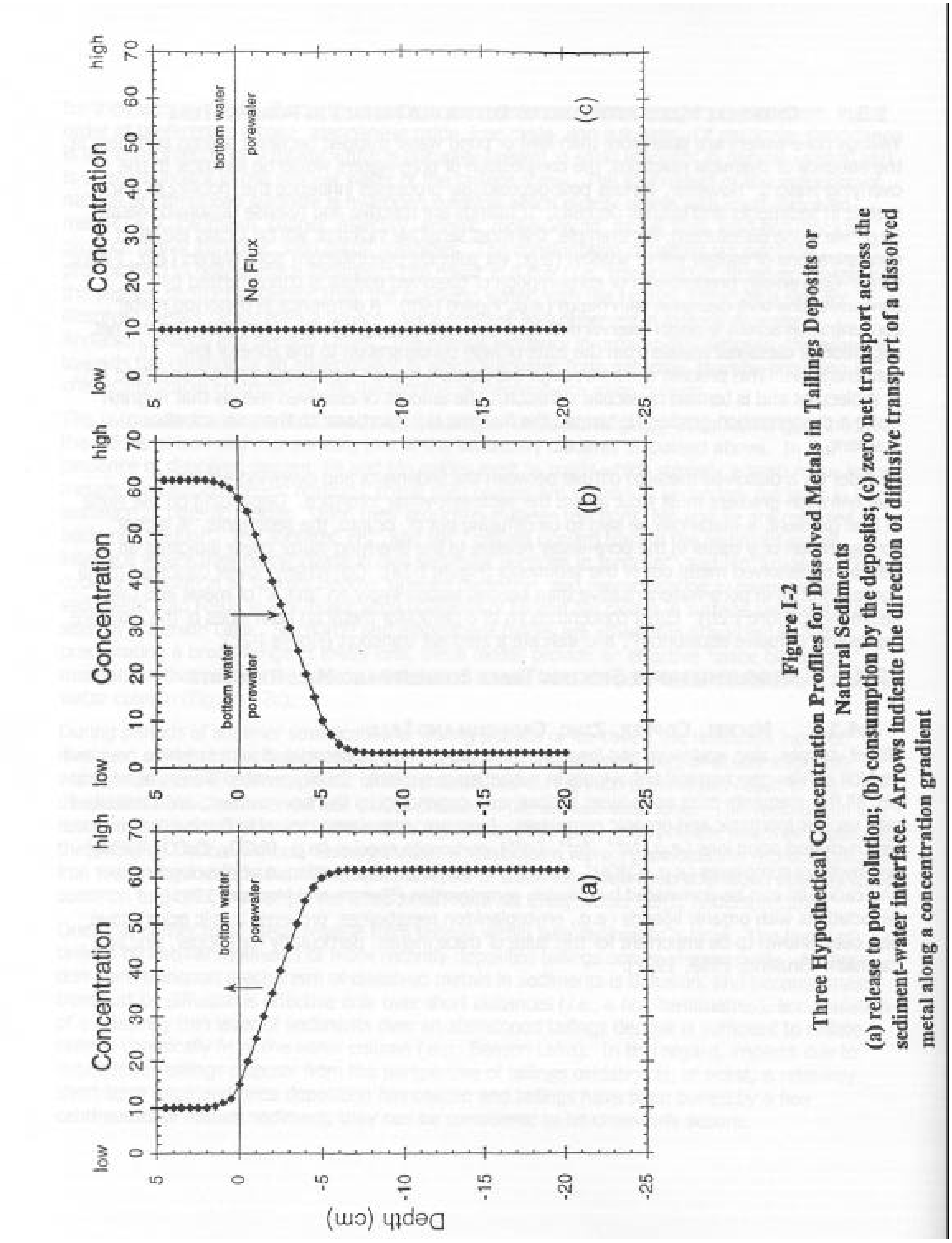


FIGURE I-2
THREE HYPOTHETICAL CONCENTRATION PROFILES FOR DISSOLVED METALS IN TAILINGS DEPOSITS OR NATURAL SEDIMENTS.
(A) RELEASE TO PORE SOLUTION; (B) CONSUMPTION BY THE DEPOSITS; (C) ZERO NET TRANSPORT ACROSS THE SEDIMENT-WATER INTERFACE. ARROWS INDICATE THE DIRECTION OF DIFFUSIVE TRANSPORT OF A DISSOLVED METAL ALONG A CONCENTRATION GRADIENT

The involvement of many trace elements in biological cycles can strongly influence their distributions in both natural systems and tailings impoundments. Metals such as zinc and copper, for example, are required micronutrients and are actively assimilated by phytoplankton (Morel and Hudson, 1985). In addition, metals can become complexed with biogenic particles via indiscriminate algal uptake and sorption to biological surfaces. Metal scavenging by organics and subsequent gravitational settling, therefore, provides an important transport mechanism of metals (as metal-organic complexes) to bottom sediments.

In the oceans, biologically important trace metals exhibit “nutrient-like” distributions, in which they are removed from surface waters by phytoplankton and eventually released back into solution at depth where detrital planktonic material is degraded by bacteria. In shallower systems such as lakes and tailings impoundments, oxidation predominantly occurs at the sediment-water interface. As a result, the degradation of organic matter often results in the remobilization of associated trace metals and subsequent release to overlying waters. Such metal recycling has been observed for nickel, copper, zinc, cadmium and lead in a variety of natural marine and lacustrine sediments (*e.g.*, Westerlund *et al.*, 1986; Morfett *et al.*, 1988; McKee, 1989).

Since organic matter can be added to tailings deposits via mixing with natural sediments such as in Anderson and Buttle Lakes, or by in situ production of algae and bacteria such as at Equity Silver, biological cycling of trace elements can be potentially important in subaqueous tailings environments. A comparison of metal effluxes from tailings and various natural deposits illustrates that the magnitudes of such remobilization processes vary substantially (Table I-1); the calculations indicate that fluxes characteristic of tailings deposits fall within the range observed for organic-rich lake sediments.

The reactivity of this suite of trace metals towards sulphide (S^{2-}) presents an additional control on their behaviour in subaqueous systems. Nickel, copper, zinc, cadmium and lead react with S2- to form insoluble sulphide mineral phases (Huerta-Diaz and Morse, 1990). The precipitation of metals as their respective sulphides (*e.g.*, CuS, ZnS, CdS and PbS) has been shown to provide an effective sink for dissolved metals in several lacustrine tailings deposits, including Mandy, Benson and Anderson lakes.

3.4.2. ARSENIC, ANTIMONY AND MOLYBDENUM

Arsenic, antimony, and molybdenum are common to sulphide ores as the sulphides arsenopyrite (FeAsS), stibnite (Sb_2S_2) and molybdenite (MoS_2), respectively. Unlike nickel, copper, zinc, cadmium and lead, this trio of elements (As, Sb and Mo) exhibits multiple oxidation states which vary according to the redox environment. Arsenic, antimony, and molybdenum typically occur in their higher oxidation states as oxyanions in oxygenated waters at neutral pH; these include arsenate ($H_2As^{V}O_4^{-}$), antimonate ($Sb^{V}(OH)_6^{-}$) and molybdate ($Mo^{VI}O_4^{2-}$). At low oxidation potentials, molybdenum is reduced from Mo^{VI} to form either MoO_2^{+} (Mo^{V}) or $MoO_2S_2^{2-}$ (Mo^{IV}). Similarly, arsenic and antimony are largely present as reduced trivalent species in reducing environments. Organic complexes such as methylated compounds also contribute to a small fraction of the total inventories of these elements in marine and lacustrine systems (Andreae and Froelich, 1984; Cullen and Reimer, 1989). Molybdenum, arsenic and antimony are also readily scavenged by iron and manganese oxides.

TABLE I-1
CALCULATED BENTHIC EFFLUXES OF DISSOLVED METALS IN VARIOUS NATURAL AND TAILINGS SEDIMENTS

Location	Sediment Type	Sedimentary Effluxes[1] ($\mu g/cm^2/y$)				
		Cu	**Zn**	**Cd**	**Pb**	**Ni**
Mandy Lake[2]	organic-rich tailings	1.0			0.8	
Buttle Lake[3]	organic-rich tailings	0.1-2.1	0.6-8.0		0.2-1.4	
Holberg Inlet[4]	tailings	0.04				
Holberg Inlet[4]	organic-rich coastal sediments	0.03				
Equatorial Pacific[5]	pelagic sediments	0.12		0.002		0.006
Eastern North Pacific[6]	hemipelagic marine sediments	0.17-0.46				
Sweden[7]	coastal marine sediments	0.27	4.5	0.053		0.45
Keogh Lake[8]	organic-rich lake sediments	0.2				
Lake Superior[9]	lake sediments	0.15			0.23	
Balmer Lake[10]	contaminated lake sediments		0.4-12			

[1] Fluxes are out of the sediments. [2] MEND, 1990a; [3] MEND, 1995b; [4] Pedersen, 1985; [5] Klinkhammer *et al.*, 1982; [6] Sawlan and Murray, 1983; [7] Westerlund *et al.*, 1986; [8] MEND 1992; [9] McKee *et al.*, 1989; [10] Martin, 1996.

Dissolved arsenic, antimony, and molybdenum are relatively abundant in the process waters of mines where molybdenite, arsenopyrite and stibnite are constituents of the ore. Molybdenite and stibnite for example, oxidize quite readily in flotation circuits, producing MoO_3 and SbO_3 which are quite soluble in water. The discharge of MoO_3-bearing tailings is consequently often accompanied by the release of dissolved Mo to receiving waters. Such releases have been observed in Alice Arm, B.C. (the Kitsault tailings), and in Rupert Inlet (Island Copper Mine tailings). In neither of these cases does the release pose an environmental risk since dissolved molybdenum is abundant in seawater and the actual amounts liberated from the tailings to the fjord waters are trivial in comparison to the natural dissolved molybdenum inventory. However, the situation could be quite different in terrestrial environments where molybdenum is not naturally abundant (*e.g.*, lakes or tailings ponds) and where, unlike seawater, the fresh waters may be used for domestic or agricultural purposes. If a natural water body is to be used as a final release point for a mine discharge where MoS_2 is a constituent in the orebody, special care should be taken to avoid molybdenite oxidation during milling. Similarly, oxidation of molybdenite-bearing waste rock should be prevented, for example by placing the rock under water immediately after initial exposure to the atmosphere.

TABLE I-2

CALCULATED BENTHIC EFFLUXES OF DISSOLVED ARSENIC IN VARIOUS NATURAL AND TAILINGS SEDIMENTS.

Location	Sediment Type	Arsenic Efflux[1] ($\mu g/cm^2/y$)
Holberg Inlet[2]	tailings	0.09
Alice Arm[2]	tailings	0.13
Equity Silver tailings pond[3]	tailings	16-80
Laurentian trough[4]	coastal sediments	0.07-0.24

[1] effluxes are out of the sediments; [2] Reimer and Thompson, 1988; [3] MEND, 1996a; [4] Belzile, 1988.

Changes in pH outside the range typical of natural waters can have a significant effect on the mobility and distribution of arsenic, antimony, and molybdenum. Antimony, for example, is least soluble at a pH of about 7, its solubility increasing at both higher and lower pH values. Thus, the discharge of lime-treated waste waters to tailings ponds might result in higher releases of Sb from sediments typically in contact with neutral waters (Mok and Wai, 1990). Although information on the environmental chemistry of antimony is limited, the data that are available suggest that the element should be least soluble in oxygenated water at near neutral pH. Similar pH considerations also apply to the solubility of arsenic. Leaching experiments have shown that As release from oxic fresh water sediments increases significantly as the pH either falls or rises from neutral (Mok and Wai, 1990). Thus, the addition of limed waste water to a tailings pond in which solid-phase, oxide-hosted arsenic is present could be expected to result in an increase of dissolved As in the pond supernatant. A comparison of measured arsenic effluxes from both tailings and natural sediments is presented in Table I-2.

The formation of metal sulphides has also been shown to influence the behaviour of arsenic, antimony, and molybdenum. In aqueous systems where dissolved sulphide is present, such as in some Canadian Shield lakes under winter ice, the formation of authigenic arsenic, antimony, and molybdenum sulphides may be anticipated. These phases should be thermodynamically stable in the absence of oxygen.

3.4.3. MERCURY

Although concentrations of mercury tend to be very low in natural waters, elevated levels are commonly associated with the mining of mercury-rich mineral deposits and industrial waste outfalls. In oxygenated waters, Hg is largely present as $Hg(OH)_2$ and $HgCl^2$ in a plus (II) oxidation state. Due to its involvement in biologically-mediated reactions, mercury also occurs in a suite of organic complexes. Methyl mercury, for example, is considered to be the most toxic form of mercury in the environment due to its high stability and ability to penetrate the membranes of living organisms (Turner, 1987). The inorganic behaviour of mercury is dominated by sorption to iron and manganese oxides. Scavenged elements, such as mercury, exhibit maximum concentrations near their sources, decreasing in concentration with distance away from inputs.

Although reports of mercury in pore-waters of natural sediments and tailings deposits are rare, some generalizations can be drawn from the available data. Associations of mercury with iron and manganese oxides, for example, were suggested to account for the input of dissolved mercury to the pore-waters in relict tailings deposits in Buttle Lake; manganese oxides have been shown to scavenge dissolved mercury readily (Lockwood and Chen, 1973). Although the Buttle deposits were characterized by some remobilization, the rescavenging of upward diffusing species in the near-surface sediments appeared to limit the migration of dissolved Hg into the overlying bottom waters. Pore-water mercury enrichments of similar magnitude were observed in mixed natural/tailings deposits in Anderson Lake (MEND, 1996b).

Like many of the trace metals discussed in this section, mercury forms insoluble sulphides in reducing sediments (Lindberg and Harriss, 1974). Although direct evidence is limited, the extremely low solubility of mercuric sulphide (HgS) suggests that dissolved sulphide should effectively precipitate mercury from solution. In some reducing sediments, the concentration of dissolved mercury has exceeded the levels predicted by sulphide equilibrium (*e.g.*, Anderson Lake). This has been attributed to the formation of soluble organic and polysulphide complexes.

The milling of sulphide ores hosting cinnabar (HgS) can result in significant enrichments of mercury in mining waste effluents. However, the extremely insoluble nature of HgS suggests that the input of such minerals should be relatively stable in subaqueous environments. The particle-reactive nature of mercury also suggests that dissolved inputs (*e.g.*, $Hg(OH)_2$) should be quickly scavenged by settling particles. Although concentrations of dissolved mercury are predicted to be low in the subaqueous setting, periodic episodes of bottom water anoxia (*e.g.*, Anderson Lake) may result in transitory releases of dissolved mercury associated with labile iron and manganese oxides.

3.4.4. CYANIDE

Cyanide is used primarily for the extraction of gold although low concentrations are also used in base metal recovery circuits as a flotation depressant. Apart from its ability to dissolve native gold and silver, cyanide dissolves other metals associated with sulphide minerals namely, cadmium, copper, cobalt, iron, lead, mercury and zinc. Cyanide species found in cyanidation solutions and tailings effluent include free cyanide, alkali-earth salts and metal-cyanide complexes with those metals listed above. The metal-cyanide complexes, particularly those that are formed with cadmium, copper, nickel, silver, and zinc are highly soluble and generally remain in the supernatant. However, tailings pore-waters can be expected to contain residual levels of cyano-metal complexes. Under alkaline conditions, dissociation of these complexes within the pore-waters would be expected to result in the release of free cyanide as CN- with the simultaneous precipitation of the respective metal hydroxide.

Some cyanide complexes are considered very stable and form insoluble precipitates that would report to the tailings solids, the most important being the hexacyanoferrates (iron cyanides). Ferrocyanide and ferricyanide can form stable salts with cadmium, copper, iron, lead, nickel, silver, and zinc, although the heavy-metal salts of ferricyanide are considerably more soluble than the ferrocyanide analogues (Smith and Mudder, 1991). Heavy-metal salts of ferrocyanide would be expected to be very stable in the tailings solids underwater. In some operations, cyanidation solutions may be treated to destroy the cyanide by oxidation with sulphur dioxide/air or hydrogen peroxide prior to deposit in the tailings pond.

II - DATA COLLECTION AND DATABASE DEVELOPMENT

1.0 INTRODUCTION

The design of impoundments with water permanently covering reactive tailings to inhibit acid generation requires extensive data on a range of physical and chemical variables. These variables help determine design criteria such as the minimum depth of the water cover, topography, meteorological conditions, and the type of structures required for permanently containing and maintaining the water cover during operations and after closure. The definition of these data and variables is a primary focus of the other parts of this design guide. Much of the discussion centres on their applicability both to the design of the physical impoundment and to the design of pond operation and closure plans and procedures for the control of water quality.

The most effective way to handle these data is to organize them in a comprehensive database that integrates chemical and physical design considerations. The database will contain all data collected in the programs that were initiated in the pre-design stages. Its ultimate use is as an iterative tool capable of indicating appropriate pond design, operating, and closure strategies for ensuring the long-term physical and chemical integrity of a tailings impoundment.

Some data are measured in field and laboratory testing programs, while other data are derived. Some data may be required to determine only one component of the overall pond design. Nevertheless, all components are interrelated and new data indicating the need for design refinements in one component will most often require a re-evaluation of the designs of other components. Hence, the process is an iterative one in which the designs of physical structures, operating procedures, and closure plans may be periodically refined or modified as new and significant information is developed and site-specific experience is gained throughout the mine life-cycle. Planned data collection and archiving programs are therefore an essential part of a tailings pond design.

The data have uses in all phases of mine development. The following is a brief overview of the types of data that will be collected not only for design purposes, but also for making risk assessments, for undertaking later analyses of potential and actual problems, and for incorporating appropriate mitigative measures.

2.0 DATA COLLECTION PROGRAMS

Data collection programs generally start as soon as a prospect shows promise of becoming a mine and will continue through each phase of development, operations, and closure. Further programs are planned and designed to satisfy the requirements of each phase in a mine development. Most of the data subsequently becomes the base for analyzing and predicting potential constraints, indicating appropriate mitigative procedures and optimizing the controllable variables to yield the final design.

2.1. PRE-OPERATIONS PHASE

The pre-operational information that can be gathered is summarized in Table II-1. Most of this information represents typical data that have been historically generated in the feasibility and design stages of mining operations. However, proper compilation and use of this dataset is essential for its application to the ongoing tailings pond management, and in particular, for predicting the short- and long-term water quality in a tailings pond.

The primary use of predicting tailings pond water quality during the pre-operations phase is incorporation of this information into an Environmental Impact Statement (EIS) which is required for the permitting of a mine. More specifically, these data could be used to:

- identify water quality parameters of concern;
- identify processes responsible for the release of contaminants (*e.g.*, oxidation, dissolution);
- predict short-term and long-term stability/reactivity of solid phases deposited in a tailings pond;
- predict whether treatment is required prior to discharge of pond water to the receiving environment; and
- derive conceptual closure/decommissioning plan for tailings pond.

2.2. OPERATIONS PHASE

During mine operations, the database should be supplemented with information recorded during the routine environmental monitoring of the tailings pond. Many of the same variables assessed or estimated in the design phase (Table II-1) should be monitored. This information would typically include tailings pond water quality, tailings mineralogy, tailings supernatant chemistry, meteorological records and hydrological data. This database could be continually used during mine operations to:

- verify or modify pre-operations/EIS predictions or pond performance;
- trouble-shoot any operational limitations/problems; and
- update tailings pond closure plan.

TABLE II-1
SUMMARY OF TYPES OF INFORMATION GENERATED FROM THE CHEMICAL AND PHYSICAL CONSIDERATIONS FOR TAILINGS POND DESIGN AND OPERATING AND CLOSURE PLANS

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

III - CHEMICAL CONSIDERATIONS FOR DESIGNING A SUBAQUEOUS TAILINGS IMPOUNDMENT

1.0 INTRODUCTION

A prediction of all inputs to a tailings storage facility is prerequisite for the effective design, management and closure of a subaqueous tailings disposal (STD) operation. The characterization and quantification of such inputs and their relative impacts is required for design purposes and demands careful consideration of several fundamental aspects common to all STD operations. These include water management, pond volume, pond residence time, mineralogy, process metallurgy, wastewater treatment and stability testwork.

The steps necessary to establish final design criteria for tailings pond operations are outlined schematically in Figure III-1. In this flow chart, several ARD mitigation options are presented. Of these only subaqueous disposal is considered here. Both physical and chemical considerations for pond design/management are integral to the physical and chemical database which, through an iterative process of design refinement and risk assessment, is used to establish a final design. The chemical considerations will be the focus of this section, with those factors that influence pond water quality after closure being emphasized.

Predicting the volume and chemical composition of various inputs to a tailings system represents a first-stage in developing operational and closure strategies. Attention must be given to the mineralogy of the ore, tailings and waste rock; milling and metallurgical processes; effluent treatment; and inputs of mine/pit water, minesite drainages and other inputs. A knowledge of these can then be used to:

- predict mass loadings of various contaminants from the respective sources;
- flag major sources of contaminants;
- identify problematic elements and/or chemical processes;
- predict the operational water quality of the tailings pond;
- establish appropriate treatment strategies;
- assess the long-term stability of tailings solids; and
- provide a preliminary approximation of post-closure water quality.

It is recognized that for many tailings ponds the water quality will be sufficiently poor during operations such that treatment will be required for discharge. However, the scope of this design guide is to ensure the long-term stability of sulphide wastes under a permanent water cover. Thus, those chemical considerations that could potentially influence pond water quality after closure are emphasized.

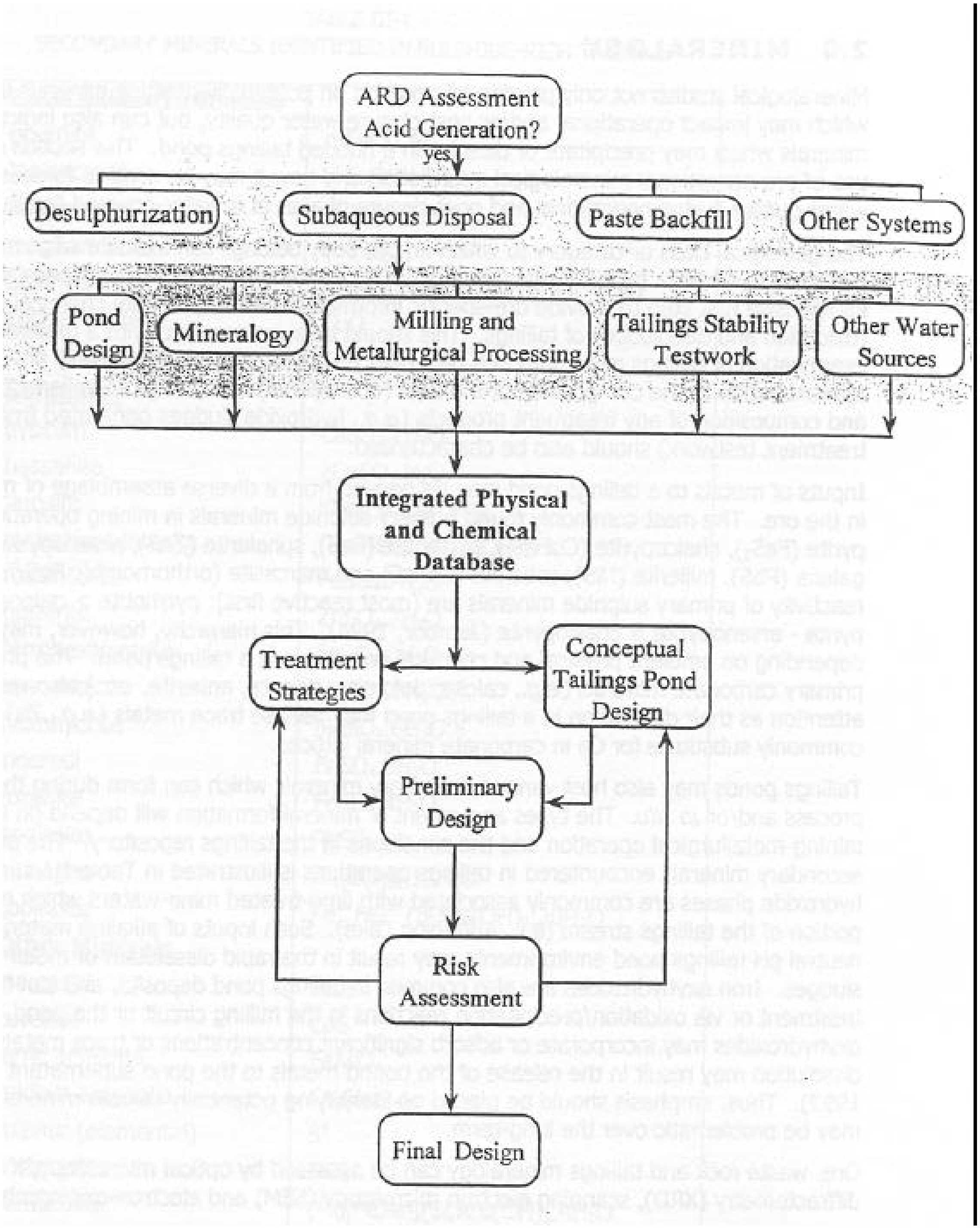


FIGURE III-1
ESTABLISHING A FINAL DESIGN FOR THE SUBAQUEOUS DISPOSAL OF SULPHIDE-RICH TAILINGS

2.0 MINERALOGY

Mineralogical studies not only provide information on potentially reactive minerals in the tailings which may impact operational and/or post-closure water quality, but can also indicate secondary minerals which may precipitate or dissolve in a flooded tailings pond. This section discusses the use of pre-operational mineralogical information and how it may be used to forecast tailings pond water quality during operational and post-closure phases of a water-covered facility.

The geological class or category to which an ore body belongs can indicate what minerals can be expected to occur in mine tailings (Kwong, 1993). A mineralogical study is therefore a rapid and inexpensive first step to provide ore-specific information on the potential chemical reactivity (oxidation and dissolution) of tailings. This should be supplemented with a mineralogical examination of tailings generated in a pilot plant, as these materials generally provide a close approximation to the composition of the final (operational) mill tailings. Similarly, the mineralogy and composition of any treatment products (*e.g.*, hydroxide sludges generated from ARD treatment testwork) should also be characterized.

Inputs of metals to a tailings pond may be derived from a diverse assemblage of minerals present in the ore. The most commonly found primary sulphide minerals in mining operations include pyrite (FeS_2), chalcopyrite ($CuFeS_2$), pyrrhotite (FeS), sphalerite (ZnS), arsenopyrite (FeAsS), galena (PbS), millerite (NiS), cobaltite (CoAsS) and marcasite (orthorhombic FeS_2). The relative reactivity of primary sulphide minerals are (most reactive first): pyrrhotite > galena - sphalerite > pyrite - arsenopyrite > chalcopyrite (Jambor, 1994). This hierarchy, however, may vary depending on ambient physical and chemical conditions in a tailings pond. The presence of primary carbonate minerals (*e.g.*, calcite, dolomite, siderite, ankerite, *etc.*) also warrants attention as their dissolution in a tailings pond may release trace metals (*e.g.*, Zn) which commonly substitute for Ca in carbonate mineral lattices.

Tailings ponds may also host various secondary minerals which can form during the milling process and/or *in situ*. The types and extent of mineral formation will depend on the particular mining-metallurgical operation and the conditions in the tailings repository. The diversity of secondary minerals encountered in tailings operations is illustrated in Table III-1. Metal hydroxide phases are commonly associated with lime-treated mine-waters which may comprise a portion of the tailings stream (e.i., Anderson Lake). Such inputs of alkaline material into more neutral pH tailings pond environments may result in the rapid dissolution of metal-bearing oxide sludges. Iron oxyhydroxides are also common in tailings pond deposits, and can form during treatment or via oxidation/precipitation reactions in the milling circuit or the pond. Iron oxyhydroxides may incorporate or adsorb significant concentrations of trace metals and their dissolution may result in the release of the bound metals to the pond supernatant (Davison, 1993). Thus, emphasis should be placed on identifying potentially soluble mineral phases which may be problematic over the long-term.

Ore, waste rock and tailings mineralogy can be assessed by optical microscopy, X-ray diffractometry (XRD), scanning electron microscopy (SEM) and electron-microprobe

TABLE III-1
SECONDARY MINERALS IDENTIFIED IN SULPHIDE-RICH TAILINGS[1]

Metal Oxides/Hydroxides	
goethite	a-$FeO(OH)$
lepidocrocite	?-$FeO(OHO)$
akaganéite	ß-$FeO(OH,Cl)$
maghemite	? -$Fe2O_3$
ferrihydrite	$5Fe_2O_3.(9H_2O)$ or $Fe_5HO_8.4H_2O$
copper oxide	CuO
zinc hydroxide	$Zn(OH)_2$ (amorph)
Sulphates	
gypsum	$CaSO_4.2H_2O$
bassanite	$2CaSO_4.H_2O$
jarosite	$KFe_3(SO_4)_2(OH)_6$
hydronium jarosite	$(H_3O)Fe_3(SO_4)_2(OH)_6$
melanterite	$FeSO_4.7H_2O$
goslarite	$ZnSO_4.7H_2O$
ferrohexahydrite	$FeSO_4.6H_2O$
epsomite	$MgSO_4.7H_2O$
hexahydrite	$MgSO_4.6H_2O$
siderotil	$FeSO_4.5H_2O$
rozenite	$FeSO_4.4H_2O$
anglesite	$PbSO_4$
alunogen	$Al_2(SO_4)_3.17H_2O$
copiapite	$Fe^{2+}Fe_4{}^{3+}(SO_4)_6(OH)_2.20H_2O$
Other Minerals	
marcasite	FeS_2
covellite	CuS
ferric arsenate	$FeAsO_4$
calcium arsenate	$Ca_3(AsO_4)_2$
sulphur (elemental)	S^o
cristobalite	SiO_2
vermiculite	$(Mg,Fe,Al)_3(Si,Al)_4(OH)_2.4H_2O$
smectite	$X_{0.3}Y_{2\text{-}3}(Si,Al)_4O_{10}(OH)_2.nH_2O$
kaolinite	$Al_2Si_2O_5(OH)_4$

---

[1] Jambor, 1994

analysis. Optical microscopy is rapid and useful for identifying sulphide minerals as well as distinguishing between polymorphs (*e.g.*, pyrite and marcasite). XRD analysis of bulk samples is used for the identification of primary minerals but is less useful for secondary mineral determinations. SEM and quantitative microprobe analyses can aid in identifying minerals which could potentially contribute trace metals to tailings pond water. For example, sphalerite can be a potential source of iron, copper, cadmium, and manganese due to the isomorphous substitution of these trace elements for zinc in the mineral lattice. Similarly, tetrahedrite $(Cu, Fe)_{12}SbS_{13}$, can be a potential source of zinc, silver, cadmium, and mercury which are often present as trace constituents in the mineral.

During the pre-operational phase, mineralogical analyses should be conducted during metallurgical testwork in order to identify sulphides and other secondary minerals within the tailings that could influence pond water quality through oxidation and dissolution processes.

3.0 MILLING AND METALLURGICAL PROCESSES

The physical and/or chemical processes employed to separate valuable minerals from the gangue component can alter the bulk mineralogy of the waste material. The following sections will examine the most commonly utilized milling and metallurgical processes with emphasis on how these can affect the chemical behaviour of the discharged tailings.

3.1. GRAVITY SEPARATION

The mineralogy of rejects or tailings produced from gravity separation should be determined in order to best manage this waste. Gravity concentration or heavy medium separation (HMS) has proven successful in liberating disseminated sulphides from lighter host rock. If the lighter material, which dominates the gravity rejects, contains significant neutralizing capacity (carbonates), some benefit could be gained by impounding such material separately from sulphide-rich tailings. Because sulphides will be managed to prevent ARD through placement underwater, gravity tailings with high neutralization capacity may better serve other mitigation or reclamation programs (*e.g.*, lining seepage collection ditches) and should therefore be kept out of the tailings pond where they offer little benefit.

3.2. FLOTATION

In addition to its primary function in treating sulphides of copper, lead and zinc, flotation as a concentrating technique has expanded to include oxides (*e.g.*, hematite, cassiterite), oxidized minerals such as malachite and cerussite, and non-metallic ores including phosphates, carbonates of non-ferrous metals, fluorite and fine coal (Wills, 1985). For the purposes of the present discussion, only sulphide flotation will be considered given the focus on prevention of ARD.

Flotation operates on the principle of altering the physical properties of the surfaces of target (or non-target) minerals. Most commonly, this entails manipulating the solution chemistry within the ore pulp in order to impart hydrophobicity (*i.e.*, water-repelling) to the mineral particles. In situations requiring more selective separation, the combined use of activators (*e.g.*, $CuSO_4$, Na_2S) and depressants (*e.g.*, lime, NaCN, $ZnSO_4$) may be necessary to enhance the selectivity and performance of collectors. These additions are extremely small in quantity and generally do not affect tailings supernatant water quality. However, to be complete, an estimate of the reagent loadings should be made in the pre-operational phase with a corresponding forecast of the pond water quality.

Because flotation circuits are generally operated under alkaline conditions, the formation of secondary metal hydroxides often accompanies the process. In some cases, lime is added prior to flotation to precipitate metal hydroxides, as this can improve the effectiveness of the collector. Xanthates can also form insoluble metal salts with ions of copper and lead. Since metal precipitates may only be stable at alkaline pH, their discharge to a more neutral pH environment in the tailings impoundment may result in dissolution and consequent elevated dissolved metal additions to the pond. These processes may persist well after closure and therefore warrant attention throughout the design and operational phases to assist in closure management.

In summary, depending on the nature of the flotation circuit, the tailings will be a mixture of solids (sulphides, gangue minerals and hydroxides) of varying degrees of solubility. Since flotation alters the overall distribution of minerals, characterization of tailings solids and supernatant will be important in terms of predicting and understanding long and short-term post-depositional reactivity.

3.3. PRESSURE OXIDATION

Pressure oxidation is the preferred technology for processing refractory gold and silver ores and concentrates from both processing and environmental perspectives. Unlike flotation tailings, pressure oxidation tailings are sulphide deplete, having been completely oxidized in the autoclaves. The oxidized slurry discharged from the autoclaves consists of an acidic liquid phase that hosts high concentrations of dissolved metals and solids that contain the gold and silver as well as hematite, basic ferric sulphates and jarosites. If arsenopyrite is associated with the ore, basic ferric arsenates can also precipitate in the autoclaves (Demopoulos and Papangelakis, 1989). While a portion of the acid liquor will be recycled, the majority of the acid overflow requires neutralization with lime prior to the gold/silver recovery processes. Autoclave residues will also report to tailings following subsequent washing, thickening and metal recovery. These barren residues are typically dominated by basic ferric sulphates, ferric arsenates, hematite and jarosites depending upon the mineral recovery process. The decision to place pressure oxidation residues in a tailings pond should be supported with adequate solubility testwork on the various solid phases in order to determine the potential impact to water quality resulting from dissolution mechanisms. In addition, such information can be used to assess possible treatment requirements and/or the need for other disposal strategies.

3.3.1. NEUTRALIZATION SOLIDS

Neutralization solids generated in pressure oxidation circuits are typically dominated by gypsum precipitates and to a lesser extent by metal hydroxides. Ferric hydroxides, for example are important scavengers of metal ions. In particular, arsenic, antimony, copper, nickel, and zinc and are very insoluble over the fairly wide pH range (5 to 8) typically encountered in a constructed impoundment. Thus, metal dissolution due to ferric hydroxide solubility is unlikely and co-precipitates would be expected to remain stable in oxic, near neutral pH tailings pond waters. However, consideration must be given to the potential for metal dissolution under anoxic conditions which may develop under the ice in the winter months in northern climates. Reductive dissolution of ferric hydroxides will result in the release of Fe^{2+} and associated metals (Cu, As, Sb, Ni, Zn) and their diffusion into the overlying water column may result in elevated dissolved metal concentrations within the impoundment. Such seasonal metal cycling mechanisms involving iron (*i.e.*, dissolution during anoxic periods and re-precipitation upon re-oxygenation of the water column) can potentially influence water quality well after closure.

3.3.2. BARREN RESIDUES

Mineralogical examination of the barren residues generated from pressure oxidation and subsequent gold recovery is important. For example, jarosite formation in the autoclaves can result in the incorporation of other metals of environmental interest including silver and lead (plumbojarosite). While this may be environmentally attractive to render the metals largely insoluble and stable, precipitation of silver jarosites may be undesirable from a metallurgical perspective. The presence of silver jarosite results in poor silver recovery and it is often necessary to perform a hot lime leach prior to the cyanide leach in order to break down the jarosites and make the silver more amenable to recovery. Other metals associated with the jarosites may also be released in such leaches and be subsequently precipitated as comparatively less stable hydroxides in the tailings stream.

Accordingly, the products of pressure oxidation circuits require careful mineralogical characterization at numerous points along the circuit to permit subsequent prediction of water quality and solids behaviour following deposition in the impoundment.

3.4. CYANIDE LEACH TAILINGS

While highly efficient in dissolving and complexing gold, cyanide also forms complexes of varying degrees of stability with heavy metals of environmental interest. It is the geochemical reactivity of these metal-cyanide complexes and the importance of characterizing cyanide-bearing waste streams that form the foundation of the following section.

Cyanide species present in gold mill effluents and circuits include free cyanide, alkali earth salts, and metal cyanide complexes with iron, cobalt, copper, nickel, cadmium, mercury, and zinc (Smith and Mudder, 1991). The tendency of these complexes to dissociate varies with the metal ion: the hexacyanoferrates (ferro- and ferri-cyanides) exhibit the greatest stability. Chemical stability in this context is defined as resistance to dissociation of the metal-cyanide complex. Degradation or dissociation of these metal cyanide complexes generally results in the release of free cyanide and the subsequent precipitation of the metal hydroxide. Sodium cyanide may also be added to sulphide flotation circuits to depress pyrite and other gangue minerals; however, discharges of cyanide from such processes are usually small in comparison to cyanide leach circuits.

Of particular importance is the tendency of ferrocyanides to form insoluble metal salts with cadmium, cobalt, copper, silver and nickel. The formation of insoluble cupro-ferrocyanide precipitates is integral to the removal of ferrocyanide from solution in both the Inco-SO_2-Air and hydrogen-peroxide cyanide treatment processes (Higgs, 1993). In addition to cyanide complexes which may not be completely destroyed or removed from the feed effluent (*e.g.*, free CN, HCN, ferri- and ferro-cyanide), treated effluents may contain various other treatment products (Table III-2).

TABLE III-2
POSSIBLE INPUTS TO A TAILINGS POND RESULTING FROM VARIOUS TREATMENT METHODS FOR CYANIDE DESTRUCTION AND RECOVERY

Treatment Method	Possible Reagents Added	Possible Inputs to Tailings Pond
SO^2-Air Oxidation	lime, copper catalyst	cyanate, thiocyanate, ammonia, hydroxide sludges, cyanide salts
Hydrogen Peroxide (H^2O^2) Oxidation	H_2O_2, lime, copper catalyst, formaldehyde	cyanate, thiocyanate, ammonia, hydroxide sludges, cyanide salts
Alkaline Chlorination	Cl_2, NaOCl, lime	chlorine, chlorinated by-products (*e.g.*, chloramines), ammonia, metal hydroxides, cyanide salts
Acidification/Volatilization/ Reneutralization	H_2SO_4,	metal hydroxides
Ion Exchange-Acidification/Volatilization/ Reneutralization	H_2SO_4	metal hydroxides

Copper will also be present in cyanide tailings as the less stable copper hydroxide ($Cu(OH)_2$) and/or cuprous cyanide (CuCN). Whether solubilized from the ore and gangue material as $Cu(CN)_3^{2-}$ or added as $CuSO_4$ to catalyze the cyanide destruction process, significant quantities of soluble copper are often associated with cyanidation effluents. Oxidation of copper-cyanide complexes releases cyanate and Cu^+, which is subsequently oxidized to Cu^{2+}. The cupric ion generally precipitates as $Cu(OH)_2$ at the high pH's typically encountered in mill effluents. Copper hydroxides can be expected to become increasingly unstable at lower pH values. This has obvious implications where the pH of the tailings pond supernatant may progressively decrease with increasing dilution from precipitation/runoff and absorption of atmospheric CO_2. The above processes will also be operative for other metal-cyanide complexes, with the majority of soluble metal complexes ultimately reporting to the solid fraction as metal hydroxides. Accordingly, the mass loading of metals associated with cyanide effluents requires quantification and should prudently be considered a future source of soluble metals should hydroxide dissolution occur.

During the operational phase, cyanide addition to a tailings pond may necessitate treatment prior to the discharge of overflow to the environment. Due to the relatively short residence time of cyanide and cyano-complexes in near-neutral pH tailings ponds, cyanide does not usually present a concern for post-closure water quality. However, the seepage of cyanide into groundwater presents a relevant post-closure consideration for permanently flooded impoundments.

4.0 CHEMICAL TREATMENT OF TAILINGS PONDS INPUTS

The treatment of tailings pond effluents is often conducted in the mill prior to discharge to the pond. Such conditions apply to cyanide-rich gold-mill effluents which may require pre-treatment to satisfy operational pond criteria, or to acidic mine-waters which may be neutralized prior to discharge. Conversely, mine/mill wastes may require treatment during and subsequent to deposition; this type of strategy, for example, may involve pH adjustments of the whole-pond environment. The last stage of mitigation may include treatment of the tailings pond over-flow prior to final discharge to the receiving environment. This is a scenario common to many mining operations where precipitation exceeds evaporation.

The following section discusses the ways in which various waste treatment systems and their products may impact pond water quality, with special emphasis placed on the treatment of acidic and cyanide-bearing waste streams. Because the complexity of water quality problems will vary considerably between mines, treatment strategies will be site-specific.

4.1. TREATMENT OF ARD

Treatment of acidic mine-waters is often required to mitigate loadings of acidity, metals, sulphate, thiosalts, and in some cases, radionuclides. Various abatement strategies may be employed, each of which may contribute a particular suite of reagents and treatment products to the tailings system (Table III-3).

The addition of lime to produce a pH range of 9.5 to 11.0 is the most commonly practiced method to both neutralize acidity and to remove metals from solution (Dinardo, 1991). Although other alkaline neutralizers may be used, the high reactivity, availability and low cost of lime make it the most prevalent (Ritcey, 1989). The neutralization/precipitation process results in the production of voluminous, hydroxide sludges which largely comprise hydrated ferrous or ferric hydroxides, gypsum, and varying amounts of co-precipitated metals (Haque, 1992). These may be discharged as low density sludges to a tailings pond or may be thickened and clarified prior to discharge.

The composition and reactivity of treated effluents may vary substantially. The post depositional reactivity of ferric and calcium arsenates, for example, has been shown to vary depending on the pH, the ratio of iron to arsenic, the concentration of calcium, and the temperature of formation (Krause and Ettel, 1985; Krause and Ettel, 1989; Harris and Monette, 1988). Due to the enhanced solubility of metal oxides at neutral pH, their disposal may degrade water quality.

TABLE III-3
POSSIBLE INPUTS TO A TAILINGS POND RESULTING FROM VARIOUS ARD TREATMENT METHODS

Treatment Method	**Reagents Added**	**Possible Inputs to Tailings Pond**
Neutralization/Precipitation	Lime, Calcite, Caustic Soda, Magnesite, Ferric Iron	Hydroxide Sludges, Sulphates, Arsenates
Sulphide Precipitation	Na_2S, $(NH_4)_2S$, BaS, H_2S	Sulphide Sludges
Solvent Extraction	Carboxylic Acids, Sulphonic Acids, Amines, Esters, Alcohols, Organo-phosphates	Organics
Sludge Flocculation	Anionic, Cationic and Neutral Polymers	Acrylamide
Thiosalt Oxidation	Chlorine, Ozone,	Cl, Sulphate, Acidity
Activated Carbon	Carbon Source	Organics

To limit metal mobility and to maximize clarity in tailings ponds, depositional strategies may include discharge of sludge with the mill tailings (Vachon, 1987). Because the volume of sludge is usually insignificant compared to the volume of tailings solids, sludge inputs tend to be sedimented and compacted more efficiently when co-disposed with the tailings. In some situations, however, maintaining high pH may be the only way to prevent sludge dissolution. Therefore, at an operation where the pH of the tailings pond cannot be reliably controlled due to such factors as variable freshwater inputs, external sources of ARD or variations in the mine/milling process, a separate sludge impoundment may be required.

5.0 OTHER INPUTS TO TAILINGS PONDS

In addition to inputs from milling operations, tailings impoundments may receive influxes from mine-site drainages, mine/pit water and natural runoff. The potential influences of these inputs on tailings ponds are discussed in the following sections.

The subaerial weathering and oxidation of sulphide minerals in waste rock can result in leachate containing significant loadings of acidity and heavy metals. At sites where such waste rock has been used for various engineering purposes (*e.g.*, fill, road construction) and/or stacked in areas around the site, the consequences can be severe. Thus, various strategies have been adopted to limit acid generation from permanently or temporarily stored waste rock (Haque, 1992; Bell, 1987) and these must be given careful consideration at the design stage. Collection and chemical treatment (*e.g.*, lime neutralization) may nevertheless be required to mitigate waste rock seepages prior to discharge to a tailings pond. The same requirements apply for acidic mine/pit waters which may be periodically pumped to the tailings area.

Minesite runoff can also contribute contaminants associated with petro-chemicals, blasting reagents and other substances. High volume natural runoff has additional implications with respect to wastewater management. In order to maintain a desired pH in an alkaline tailings pond, for example, additional buffering may be required during periods of elevated inputs of near-neutral pH water. Furthermore, large volumes of tailings supernatant overflow may be present where fresh water inputs are excessive. During operations, such inputs should be limited by diversions or other means in order to minimize the water handling and treatment requirements. Following closure, however, natural run-off directed to the pond may help to maintain the water cover.

5.1. CONTAMINANT MASS BALANCE

The relative role of each flow into the tailings pond can be assessed by preparing a contaminant mass balance model. This model consists of compiling flow-rate information and water-quality data from each flow stream. These data can be obtained from estimations from laboratory testing or measurements from the field programs. Wherever possible, a range of data should be used to reflect a range of conditions such as would be experienced in seasonal runoff. By combining the flow rate with concentration data, a mass loading rate is obtained which concisely illustrates the relative contribution of each stream to the overall contaminants in the pond. Examples of a typical mass balance are developed in Appendix D.

This information can be used to optimize the tailings pond management for both short-term and closure considerations. These may include the following options:

- determine the need to pre-treat flows before discharge into to the pond;
- manage some streams independently of the tailings system pond;
- optimize the type and size of treatment plant (if needed for discharge);
- optimize the piping systems to the pond and/or other areas of the project.

IV - PHYSICAL CONSIDERATIONS FOR TAILINGS POND DESIGN

1.0 INTRODUCTION

Most of the data describing physical design variables are site-specific, depending on factors such as foundation permeability and mining and milling processes for a particular ore. Hence, the primary purpose of this part of the design guide is to define the variables and suggest criteria adaptable to a broad range of pond designs. The intent is to provide a framework which can be adapted to site conditions and refined with precise data gathered during the design, operating, and closure phases of a mine life cycle. The focus is on constructed or man-made tailings ponds.

Note that designing a tailings pond using water cover as an acid generation control technique assumes that the method of deposition permits distribution of the tailings within the pond such that beaches are not developed. Also assumed is a method of deposition that does not allow the development of subaqueous slides.

2.0 WATER COVER DESIGN OBJECTIVE

The water cover design objective is to provide an adequate depth of water to ensure the consolidated bed of tailings is not entrained or remobilized during operation and after closure of the pond. The establishment of a physically stable tailing mass is required for the chemical profiles described in section I 3.3 to develop these metal gradients effectively and limit metal mobility out of the tailing.

During operation of a tailings pond the bed is likely to be comprised of a layer of unconsolidated material overlying a more consolidated layer (Figure IV-1). The upper layer acts more as a fluid than as a bedded layer of tailings until settlement and initial consolidation takes place.

3.0 PHYSICAL PROCESSES AFFECTING BED STABILITY

Two principal factors affect bed stability under a body of water. These are (1) the entrainment forces imposed on the sediment surface and (2) the resistance of the sediment to entrainment. Several physical processes are capable of exerting sufficient force to overcome the resistance of the sediments to entrainment. These processes will be outlined in the following sections.

The entrainment of sediment from the bed may be the result of shear stress imposed on the bed surface by movement of the adjacent water. Waves and currents cause water to move above the bed and this movement of water creates a zone of shear at the water-sediment interface. The shear imparted on the bed is related to the speed with which the water moves. Shear stress increases as the square of velocity. Typically, the strength of wave-induced currents are related to the length and depth of the water body, such that the wave-induced bottom velocity will increase with increasing tailings pond size. For this design guide, the assumption is that the method of tailings deposition is such that currents associated with the discharge of tailings are confined to the proximity of the outfall.

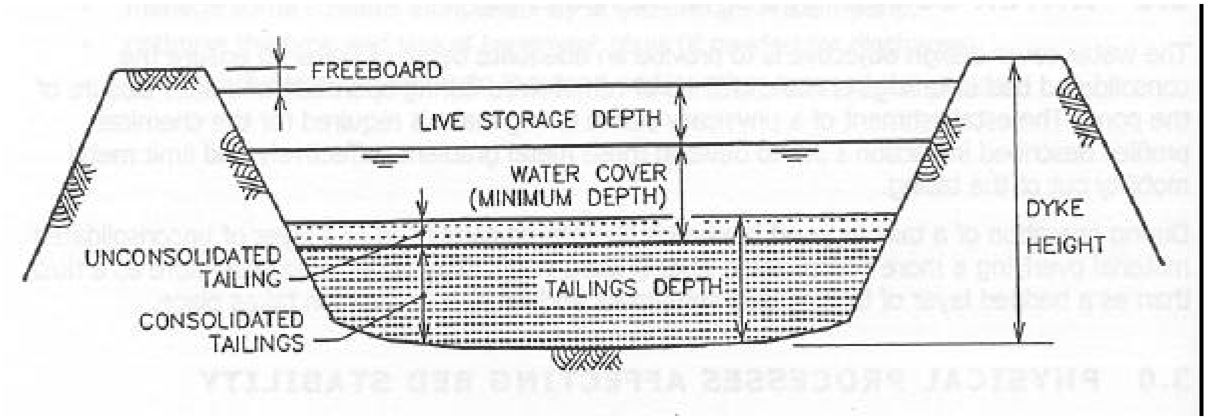


FIGURE IV-1
TAILINGS POND CROSS-SECTION

Entrainment of sediment from the bed may also be the result of physical removal of sediments by contact with ice. Proximity of ice to the bed may cause sediment grains to freeze to the overlying ice mass. Ice entrainment will only occur where the ice actually comes into contact with the bed.

The resistance of sediments to entrainment is a function of inherent sediment properties. The critical shear stress at the bed required to entrain sediment may be considered as a function of particle size and density. Larger, heavier sediment grains will tend to require greater shear stresses in order to be entrained.

Like the strength of currents, the physical processes which affect bed stability within an impoundment will vary relative to the overall impoundment geometry and size. The surface area of the impoundment as well as its mean depth will have a significant impact upon the importance of various processes. These are divided into those which would likely have little impact on small constructed ponds and those which will play a significant role. This division may be made on the basis of relative pond size.

3.1. PROCESSES WITH MINIMAL IMPACT

When determining what physical processes may affect a specific impoundment, the physical size of the impoundment needs to be assumed. The spatial extent of an impoundment will influence to some degree which processes are important to bed stability. To this extent, water bodies may be classified by area into small (<5 km^2), medium (5-10 km^2) and large (>10 km^2) surface areas. These may then be sub-divided into shallow (mean depth <5 m), moderately deep (mean depth 5-10 m) and deep (mean depth >10 m) water impoundments.

Since the physical processes affecting sediment stability are dependent upon site geometry, larger or smaller ponds will be affected by different processes (Table IV-1). In small, constructed ponds, where depths are often less than 10 m and areas are often less than 5 km^2, the processes which affect the larger, deeper water bodies normally have relatively negligible impact on bed stability.

For example, seasonal turnover, caused by seasonal heating and cooling of the impoundment water, may create currents which carry sediments to the surface as the water upwells. Whether seasonal turnover occurs, or whether daily turnover occurs, is a function of whether thermal stratification occurs in an impoundment. Gorham and Boyce (1989) have shown that thermal stratification is a function of lake size and area. Generally lakes shallower than 3 m remain unstratified, whereas if 5 m, or 10 m, deep, the lake would remain usually unstratified if open-water lengths exceeded about 200 m and 500 m, respectively. It is assumed that for small water bodies with depths less than 5 m that wave-induced bottom currents will be greater than currents associated with turnover.

Also wind-driven build-up of water at one end of a water body which results in water seiching back-and-forth when the wind dies, may generate currents which entrain sediments. However, it is only a process of significance on the largest of water bodies where fetch lengths are of the order of tens of kilometres. Small, man-made impoundments, with fetch lengths of the order of kilometres, will likely be too small to experience significant seiching, thus this process will likely not affect bed stability. Similarly, currents associated with density-driven internal waves are influenced by thermal stratification which, as noted above, are assumed to be of less significance than wave-induced currents on small man-made impoundments.

Currents formed by inflow of water, longshore generated wave motions, and/or thermally driven recirculation may entrain and move sediments. If the current velocity is large enough to generate sufficient shear stresses at the bed, the current will then entrain material. In large water bodies, with surface areas of the order of hundreds of square kilometres, these residual currents may be large. In small, constructed impoundments, with surface areas of the order of square kilometres, with no inflow or outflow other than pumped discharges, these residual currents will likely have low velocities and low shear stresses and will likely not affect bed stability.

TABLE IV-1
PROCESSES AFFECTING BED STABILITY

Water Body Area	**Small** **< 5 km^2**			**Medium** **5 - 10 km^2**			**Large** **> 10 km^2**		
Water Depth (m)	0 to 5	5 to 10	10 +	0 to 5	5 to 10	10 +	0 to 5	5 to 10	10 +
Physical Process									
Seiching							●	●	●
Seasonal turnover			●			●			●
Currents				●	●	●	●	●	●
Wave action	●	●	●	●	●	●	●	●	●
Ice entrainment	●			●			●		

As shown in Table IV-1, wave action and ice entrainment are the principle processes that may affect bed stability in small impoundments. These two processes, together with sediment properties, will be further detailed in the following sections. The assumption that wave-induced currents will control the design of water cover depths over other sources of currents should be verified on a site-specific basis.

It is further assumed, within the context of this design outline, that the method of tailings deposition is such that currents associated with the discharge of tailings are confined to the proximity of the outfall.

3.2. WAVE ACTION

Wave entrainment of sediments occurs when waves are large enough that the velocities associated with the passage of the wave are strong enough to mobilize material at the bed, Figure IV-2. Sediments entrained by the wave motion are carried into the water column by the efficient stirring action of the wave. Sediments entrained by wave action may be carried high into the water column and erosion of the bed may take place if the wave-entrainment of material is extensive.

Wave-action entrainment will occur where the water is shallow enough such that the velocities associated with the passage of the wave are greater than the critical velocities required to entrain sediment. For a given wave height, wave-induced motion at the bed decreases with increasing water depth. At some critical depth, no entrainment of sediment occurs.

Key variables influencing wave action entrainment of sediment include fetch length, windspeed, wind direction and sediment properties such as grain size and density. Bathymetry may influence waves by modifying deep water waves as they enter shallow water. Additional parameters include water density and wave height coefficients. Variations in the elevation of the water surface as a function of the water balance, calculated as part of pond design, may also influence the effect of wave action.

The critical depth may be predicted by calculating the depth at which wave-induced velocities fall below critical values. These depths are dependant on the characteristic wave height and wave period determined from wave hindcasting using wind data.

3.3. ICE ENTRAINMENT

In areas which experience winter temperatures below freezing, ice entrainment occurs when the ice cover on the pond grows to a thickness sufficient for the ice to be grounded on the bed. Sediments may freeze to the ice as a result of the ice-sediment contact and may be floated from the bed during the spring thaw. As the ice melts, the sediments entrained in the ice are released into the water column. Where mean water depths are greater than 5 m, seasonal freeze-up is unlikely to generate an ice cover that is thick enough to reach the bed.

The key variables affecting ice-entrainment include water temperature, air temperature and the number of days below freezing. Other parameters include heat capacity and heat conductivity of the air, water and ice phases as well as the heat exchange coefficients across the air-ice and ice-water interfaces.

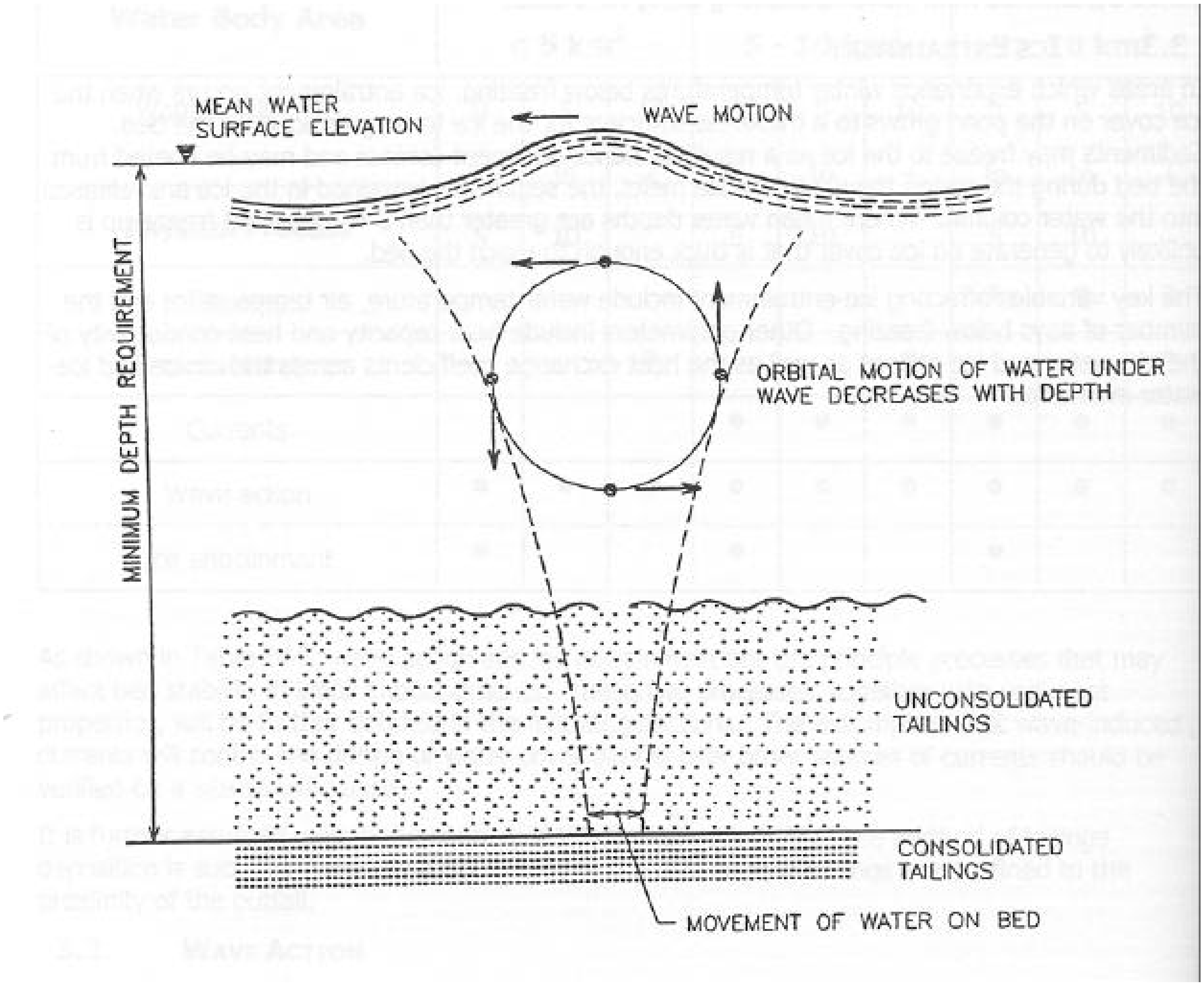


FIGURE IV-2
WAVE INTERACTION

The critical water depth may be predicted by calculating the ice growth thickness as a result of winter freezing. These depths are dependant on the thermal gradients and diffusion coefficients of the surrounding water and air. Other variables which are difficult to quantify but may affect the thickness of the ice are the depth and insulation of snowfall, ablation and drifting of snow onto the ice.

3.4. SEDIMENT PROPERTIES

Sediment properties can have a significant influence on the entrainment process. The key sediment properties affecting resistance to entrainment are grain size and sediment density. Grain shape may also influence entrainment by packing or locking particles together more effectively. Large particle, heavy sediment requires large shear stresses to overcome the force of gravity and be entrained away from the bed. Small, clay sized particles require large shear stresses to overcome the grain-to-grain cohesiveness of small particles caused by electrostatic attraction.

The key variables associated with wave action, ice entrainment and sediment properties are summarized in Table IV-2 which outline a comprehensive data collection program designed to address the data needs of minimum depth criteria. Some additional variables are included in the data collection program from the point of view of pond design. These pond design variables and their inter-relationships with the depth criterion variables are presented in the following section.

TABLE IV-2
KEY VARIABLES AFFECTING MINIMUM DEPTH CRITERIA

Process	Measured	Derived	Output
Wave Activity	wind speed wind direction fetch tailings grain size tailings density	wind duration wave period wave height	bed velocity minimum depth
Ice Entrainment	water temperature air temperature relative humidity	ice duration days below 0^{o}C	ice thickness minimum depth
Sediment Properties (Tailings)	sediment weight sediment volume sediment size	sediment density	

4.0 MINIMUM DEPTH CRITERIA

A minimum depth of water for a pond needs to be designed to ensure that the sediments on the bed are not affected by physical processes. In small, constructed impoundments this criteria is based on the maximum of:

- minimum depth to protect from wave activity; and,
- minimum depth to protect from ice entrainment.

4.1. WAVE ACTIVITY

To protect the bed sediments from wave activity, a minimum water depth criteria may be predicted using a method developed by Lawrence *et al.* (1991) coupled with a critical bed velocity computation derived from the work of Komar and Miller (1975a,b).

The computation procedure requires the solution of the three separate equations presented below. The input variables required for these calculations are:

- D_{50} median grain size of bed sediments ,mm;
- F fetch length, m;
- D_s sediment density; and,
- U_w windspeed for the site, ms^{-1}, where $U_w = 0.71\ U^{1.23}$.

Calculated variables (with computed dimensions/units) are:

- T_{sig} significant wave period, s (average period of the highest 1/3 of all the waves in the wave spectrum);
- V_b critical bed velocity, m/s; and,
- d_{min} minimum required water depth, m.

The significant wave period for wind-generated waves may be computed using the equations developed by the US Army Coastal Engineering Research Center (1984). These wave periods may be calculated for both deep water and shallow water waves. The equation to calculate the significant wave period, T_{sig}, for deep water wave is:

$$T_{sig} = 5.93x10^{-2} (U_w F)^{\frac{1}{3}}$$

Using the result from this equation, the critical velocity at the bed may be estimated from the work of Komar and Miller (1975a,b). The equation to calculate the critical bed velocity, V_b , for these waves is:

$$V_b = (0.21(\frac{\rho_s - \rho}{\rho}) g (\frac{D_{50} T_{sig}}{\pi})^{\frac{1}{2}})^{\frac{2}{3}}$$

where the additional constants in the equation are:

- D water density, kg/m^3; and,
- g gravity, m/s^2.

Using the result from this equation, the minimum depth may be calculated according to the method of Lawrence *et al.* (1991). The equation to calculate the minimum depth requirement, $d_{\min}$, is:

$$d_{\min} = 0.00187 \left(\frac{U_w^{\,2} F^2}{g} \right)^{\frac{1}{3}} \ln \left[0.037 \frac{R (g F U_w^{\,4})^{\frac{1}{6}}}{V_b} \right]$$

where the additional variable is:

- R a wave height coefficient (for H_{sig} , R=1).

The calculated minimum water depth may then be applied to impoundment design and used as a guideline in water balance calculations. The pond design should ensure that the minimum depth is maintained throughout the hydrologic year as part of the water balance calculations for design purposes.

The complete derivation of this minimum depth calculation procedure is presented with references in Appendix D, noting that implicit in the above equations is the assumption that deep water waves are generated within the impoundment; an assumption that requires checking in the final design.

4.2. ICE ENTRAINMENT

To determine a minimum depth requirement to protect the bed sediments from entrainment by ice requires the modeling of ice growth associated with the freezing of the pond. Models for ice growth do exist but they are fairly complicated and require an iterative procedure to determine the maximum ice growth during the winter season.

These models rely on the thermal exchange properties between air, ice and water to calculate the depth of water through time. The rate at which ice growth occurs is controlled by several factors which include:

- the thermal gradient across the air-water and later air-ice, ice-water boundaries;
- the latent heat of fusion of water;
- the thermal conductivity of the air, water and ice;
- ablation of the ice surface; and
- the vapour pressure/relative humidity of the overlying air.

A simple formulation of the thermal exchange equation is beyond the scope of this manual, but ice entrainment should be recognized where the potential for such a process exists. The sensitivity of the result to temperature fluctuations may dictate that a safety factor be added to the ice thickness depth to ensure no ice grounding occurs.

The recommendation for the minimum water cover in a pond is that it should be at least 10% deeper than the maximum ice thickness at any time during the winter months.

5.0 DESIGN PROCEDURE OUTLINE

A logic chart for the design of a tailings pond with water cover is shown in Figure IV-3. The chart identifies four procedural loops which in a reiterative process can achieve an economical design that meets environmental criteria at an acceptable level of risk. The four loops are:

- Impoundment Area Variation. The initial estimate made for pond dimensions may be based on tailings throughput, pond life and topographic conditions. The pond water cover depth is a function of the length and depth of the pond. The economic analysis of the best choice of pond width and depth requires a number of reiterations in order to generate costs for each option. Dam location is dependent on natural topography.

- Design Event Variation. The capital and maintenance costs associated with elements of a tailings pond, such as dyke heights, are a function of the severity, or probability, of the design event. The design event may be related to wind, precipitation, or other climatic or seismic events. In cases where the severity of the design event is not predicated by regulatory agencies, the most economic design is developed through reiterations in which the intensity of the design event is varied.

- Storage Volume Variation. The volume of water available between the minimum water cover and encroachment on the freeboard of the dyke is the live storage of the pond,

Figure IV-1. The amount of live storage required is a function of the pumping capacity installed for the operation of the pond. For given pond dimensions and precipitation intensity there is an optimum combination of dyke heights and pump capacity that is determined by reiterative calculations.

- Tailings Process Variation. Minimum water cover and impoundment volume are a function of the size and throughput volumes of tailings. A change in either of these variables may change the most economic configuration of the tailings pond.

Individual design steps are discussed briefly as follows:

5.1. STEP 1: COLLECT EXISTING FIELD DATA

All available existing site field data concerning the variables relevant to bed stability and pond design should be compiled. These data may be available through government agencies such as environmental or natural resource departments or through companies with previous development projects in the area. The data which should be collected includes climatic, hydrologic and sedimentologic variables.

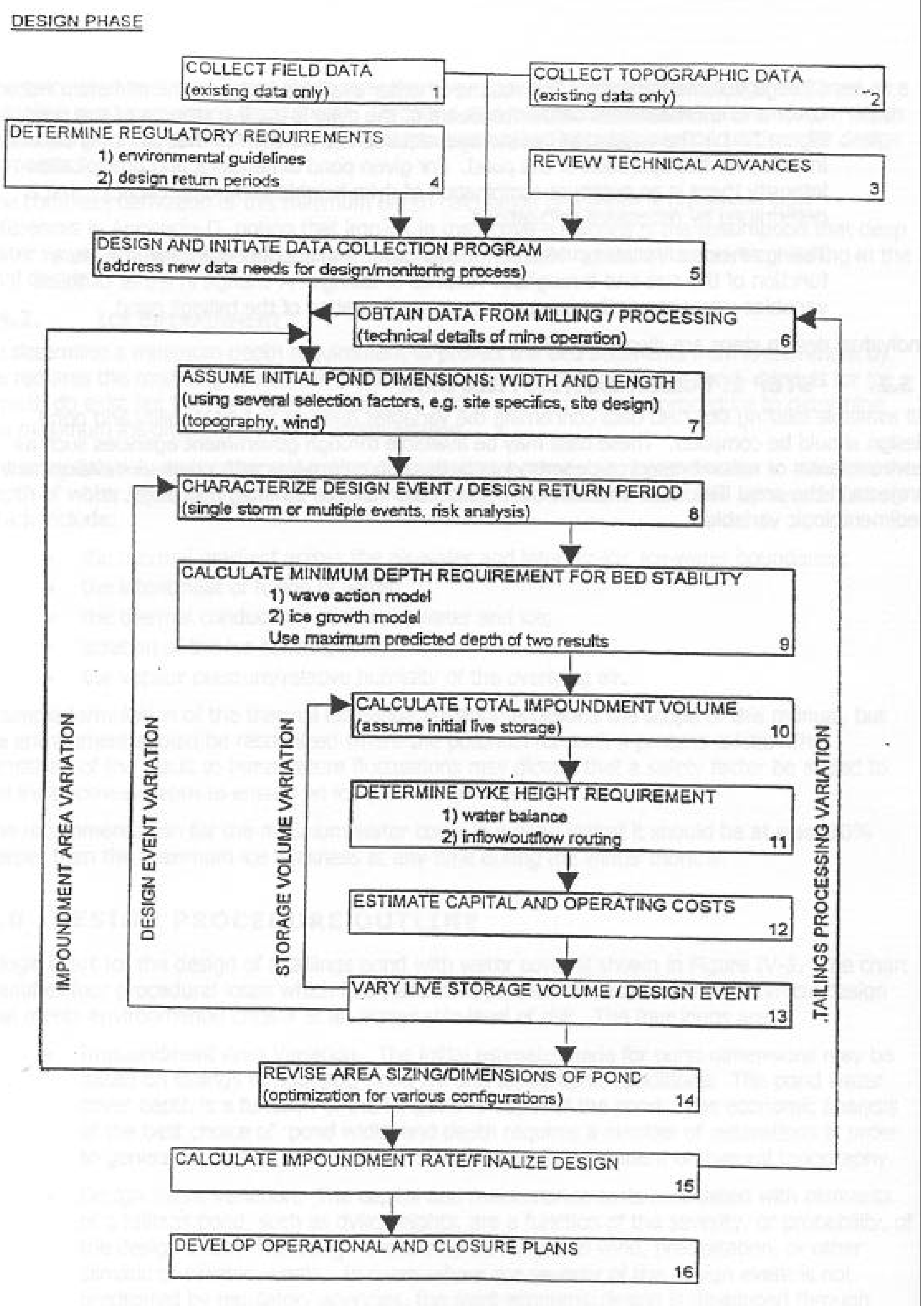


FIGURE IV-3 DESIGN LOGIC CHART

5.2. STEP 2: COLLECT TOPOGRAPHIC DATA

Existing topographic information covering the site should be obtained. These data will serve as the baseline topography for the site. The elevations provided may be used to help pinpoint and prioritize areas requiring detailed surveys. These data may be available in the following formats from government agencies or commercial map and airphoto distributors:

- topographic maps;
- previously detailed surveys;
- aerial photographic stereo-pairs;
- airborne imagery; and,
- satellite imagery.

The compiled topographic data should be used to delineate the primary arterial streams and rivers crossing the site and their associated drainage basins, drainage areas, elevation differences within each basin and basin orientation. Additionally, all compiled field data could be geo-referenced on a map or in a Geographic Information System (GIS), and recorded relative to the basins and streams.

Site surveys will likely be required to fill in gaps in existing topographic data. Consideration should be given to formatting topographic data into a Digital Terrain Model (DTM) of the site allowing for direct translation of elevations into computer design (CAD) software packages and the derivation of volumes such as cut and fill quantities associated with earthworks.

5.3. STEP 3: REVIEW TECHNICAL ADVANCES

A review of relevant technical literature should be completed and should cover technical, environmental, and scientific journals as well as governmental and industry reports related to the location of the site. The review should identify any recent advances in technical developments, scientific knowledge, construction methods and environmental considerations. If any advances are identified which could improve the site's efficiency, stability, operating costs and environmental management, these should be incorporated into the design process where possible.

5.4. STEP 4: DETERMINE REGULATORY REQUIREMENTS

Prior to the initiation of design work, the applicable regulations and objectives established by the appropriate regulatory bodies should be determined, keeping in mind that regulations may not be available for every design component. Typical regulations affecting the design of a tailings pond include:

- water quality objectives for the effluent discharge; and,
- acceptable levels of risk with regard to non-compliance with environmental standards.

In conjunction with the expected life of the tailings pond the acceptable levels of risk will establish the design events or the design storms, as indicated in Figure IV-4.

5.5. STEP 5: DESIGN AND INITIATE DATA COLLECTION PROGRAMS

Following the initial gathering of existing data, a data collection program should be developed to fill any gaps in the data set. A field data collection program should be initiated to supplement existing data from step 1, recognizing that data collection will likely carry on throughout the operation of the tailings pond and in to the post-closure phase to augment the initial database and verify design and operational decisions made based upon that database.

The data collection program should reflect both short-term requirements for design and the longer term requirements for successful operation and closure of the pond.

5.6. STEP 6: OBTAIN DATA FROM MILLING/PROCESSING

The data required from the milling and plant design include:

- tailings throughput, t/d;
- processing discharge to the tailings pond, m^3/s;
- bulk density of tailings, t/m^3;
- grain size distribution of tailings and,
- runoff, minewater and/or other flows to the tailings pond, m^3/s.

5.7. STEP 7: ASSUME INITIAL POND DIMENSIONS (WIDTH AND LENGTH)

The initial and final pond dimensions are a function of the pond operating and expansion strategy. Impoundment capacity throughout the life of the mine may be developed by increasing the height of containment dykes, adding additional containment cells, or a combination of both. Therefore to initiate the design loop that deals with impoundment area, it is necessary to assume an initial containment volume.

By way of example, assume that the initial pond, prior to expansion, will be capable of containing five years of tailings production within an initial surface area of 1 km^2. Based on this assumption a pond length and width are then chosen recognizing that several possible options may be available for the impoundment location and geometry. Factors to consider in the initial and subsequent selection of pond dimensions and location include:

- the depth of water cover required will likely be reduced if the shortest dimension of the pond is aligned with the strongest winds;
- conversely, the major axis of the pond would best be aligned in the direction of the lightest winds and,
- capital costs will be reduced if advantage is taken of natural topographic depressions or ridges, while minimizing the requirement to divert existing major drainage courses.

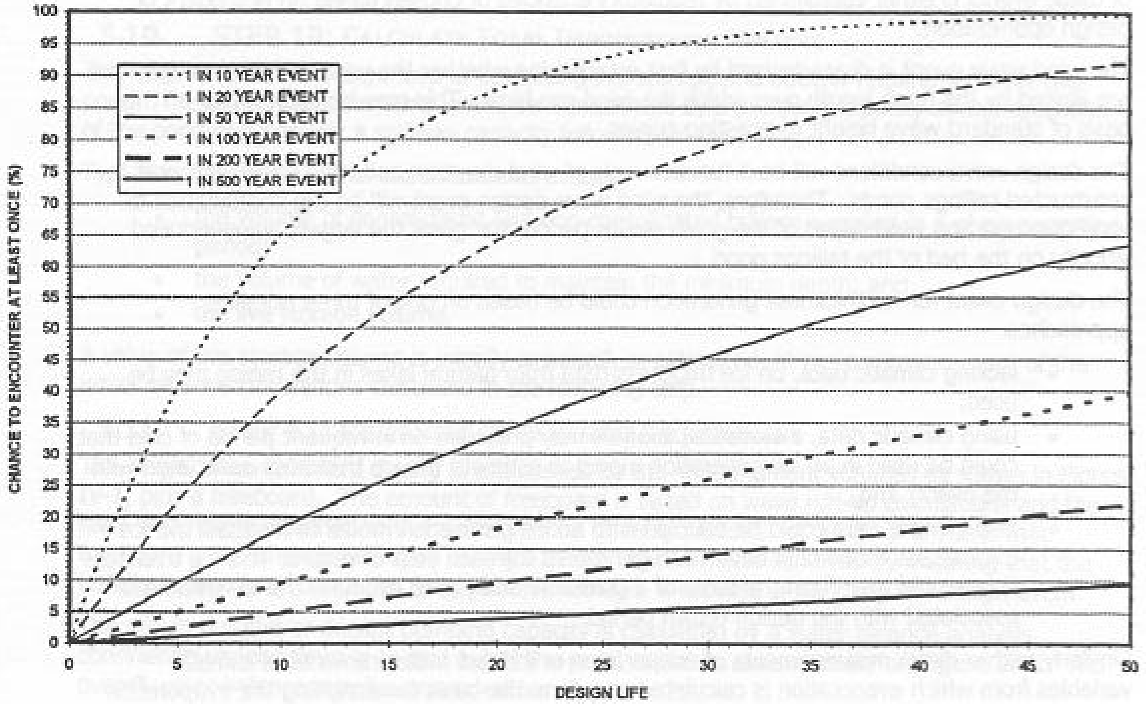


FIGURE IV-4
RECURRENCE INTERVAL RISK

5.8. STEP 8: CHARACTERIZE DESIGN EVENT/DESIGN RETURN PERIOD

Apart from seismic events, four basic design events need to be considered:

- wind wave generation as it affects water cover and dyke crest elevations;
- prolonged periods of cold as it affects ice development and water cover requirements;
- prolonged periods of drought as they affect evaporation and make up water requirements to maintain adequate water cover; and,
- precipitation as it affects the water balance, live storage requirements, and possibly stream diversion.

As previously mentioned, the level of risk as characterized by the return period assigned to each of these events is either established by regulatory agencies or chosen based on precedent or design optimization.

The wind wave event is characterized by first establishing whether the wave heights on the pond are limited by the fetch length over which the wind can blow. This may be established on the basis of standard wave height forecasting curves.

The design wave conditions will be a function only of wind direction and wind speed for most constructed tailings ponds. Therefore, the wind wave design event will be the combination of wind direction and wind speed of the given return period that gives the largest wave-induced velocity on the bed of the tailings pond.

The design event for ice thickness generation could be based on one of three possible approaches:

- lacking climatic data, on ice thickness data from natural lakes in the region may be used;
- using climatic data, a statistical analysis may prescribe an infrequent period of cold that could be used in an ice generation model to estimate the ice thickness associated with the event; and,
- also climatic data could be coupled with an ice generation model to hindcast the ice thicknesses that would have been generated for each year of record. This ice thickness data would then form the basis of a statistical analysis to estimate the ice thickness associated with the design return period.

Similarly, either direct measurements of evaporation or indirect measurements of climatic variables from which evaporation is calculated may form the basis for assigning the evaporation design event of a given return period.

The design events for precipitation are derived from precipitation data that includes snowfall as a water equivalent. The volume and rate of direct inflow into the pond is a function of both precipitation intensity and duration. Various combinations of intensity-duration events of the same probability, or return period, are required for input into a water balance analysis. Multiple events, of equal probabilities to those of the single events, may also be required to assess the critical combination of intensity and duration.

Design events for runoff associated with the design of drainage around the pond may be derived from precipitation data which is often summarized as intensity-duration-frequency (IDF) curves.

Standard probability theory and extreme value analyses are normally applied to climatic data to derive design events of given average return periods.

5.9. STEP 9: CALCULATE MINIMUM DEPTH REQUIREMENT FOR BED STABILITY

Evaporation and seepage losses need to be considered when the minimum depth is established by wind wave conditions. If make-up water is not provided then the water cover should be the minimum depth of water left in the impoundment at the end of the dry season following a drought year of a set return period, Step 8, such that:

$$d_{min} = d_{crit} + ?d_{evap} + ?d_{seep}$$

where, d_{min} is the water cover to maintain in the pond, d_{crit} is the depth of water calculated to eliminate entrainment of tailings by wave action, $?d_{evap}$ is the change in water level due to evaporation during a prolonged dry spell, and $?d_{seep}$ is the drop in water level due to seepage losses over the same dry spell.

5.10. STEP 10: CALCULATE TOTAL IMPOUNDMENT VOLUME

This step initiates the design loop of optimizing live storage and outflow requirements for the pond. The outflow requirements would be represented by a pumping capacity during operation of the pond and possibly a spillway capacity following closure.

Typical tailings pond within impoundments include;

- the volume of consolidated and non-consolidated tailings supplied in a given operating period;
- the volume of water required to maintain the minimum depth; and,
- the live storage volume.

A value of live storage volume is initially assumed, or estimated, at the commencement of the water balance calculations discussed in the following step.

5.11. STEP 11: DETERMINE DYKE HEIGHT REQUIREMENT

The height of containment dyking is established by the impoundment volumes as shown in Figure IV-1, plus a freeboard. The amount of freeboard is based on wave run-up calculations and the level of certainty attached to the determination of design events. A design premise is that freeboard is not encroached upon during a design event. This is achieved by ensuring that the installed capacity of the outlet flow, plus live storage, is capable of handling the design event.

The required outlet or decant pumping capacity is calculated by a water balance analysis considering various flows in to and out of the pond as shown schematically on Figure IV-5. The overall water balance may be expressed as:

$$?S = P + T + M - (D + S + E)$$

where ?S represents the required live storage, P is the volume of precipitation reaching the impoundment surface, E is the evaporation volume from the surface, M is the pumped make-up water to maintain the minimum depth criteria, D is pumped decant water to maintain the live storage needed, S the seepage losses and T is the water from the tailings slurry pumped into the pond.

Assuming operation during a design precipitation event coupled with tailings flow, to size the decant pump capacity requires an inflow-outflow routing calculation. Starting with all live storage available, the size of decant pump required is that which makes full use of the live storage available before water levels in the pond commence to recede. Evaporation and seepage losses are often not required to be included in this calculation because they are small relative to inflows. Completion of the routing calculation then links a dyke height with a required decant pumping capacity for a given live storage. A larger value of live storage will increase dyke elevations but decrease the size of pumps required.

Note that a reduction of the live storage required by utilizing the decant or outflow capacity is viable only if such outflow is reliable during storm events. Otherwise live storage should be designed to handle the entire precipitation from the design event. Note also that external factors, such as exposure to flooding, may also affect the height of a containment dyke, being a consideration for overall plant layout and design.

5.12. STEP 12: ESTIMATE CAPITAL OPERATING COSTS

Given an impoundment area, dyke height and pumping capacity, a benchmark estimate of the capital and operating costs should be prepared. This estimate should include any other cost components that are affected by the pond size, decant pump size, and/or dyke height in order to make fair cost comparisons between alternatives. Completion of the cost estimate at this point permits design reiterations to be made.

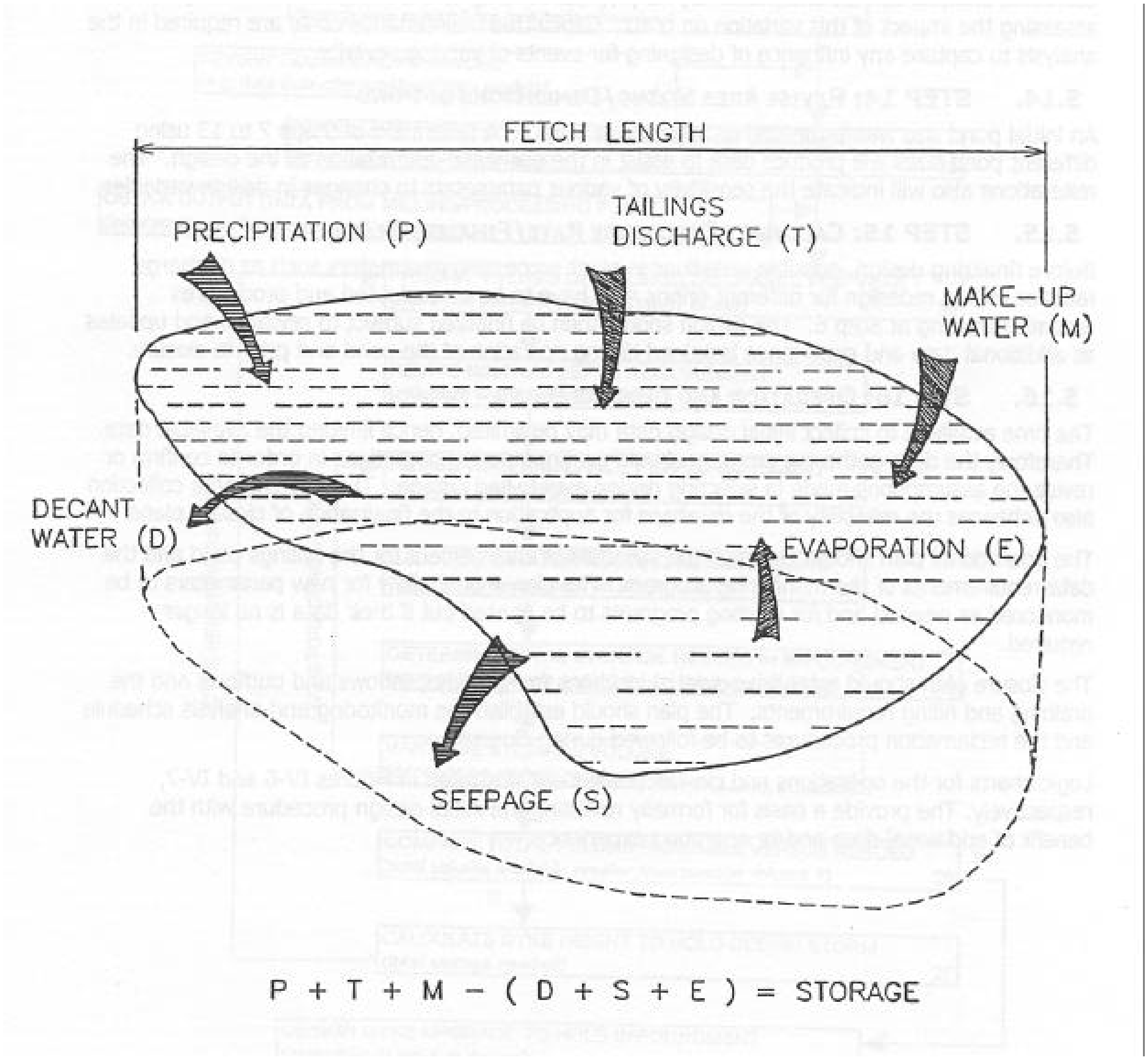


FIGURE IV-5
WATER BALANCE

5.13. STEP 13: VARY LIVE STORAGE VOLUME/DESIGN EVENT

The live storage initially assumed in Step 10 should be changed and Steps 11 and 12 repeated if economic optimization is required for the installed pump capacity.

Similarly, if the probabilities of all the design events are not established by regulation, then an economic analysis can be undertaken by varying the severity of the design event at Step 8 and assessing the impact of this variation on costs. Capitalized maintenance costs are required in the analysis to capture any influence of designing for events of varying severity.

5.14. STEP 14: REVISE AREA SIZING/DIMENSIONS OF POND

An initial pond size was estimated or assumed in Step 7. A reiteration of Steps 7 to 13 using different pond sizes will produce data to assist in the economic optimization of the design. The reiterations also will indicate the sensitivity of various parameters to changes in design variables.

5.15. STEP 15: CALCULATE DISCHARGE RATE/FINALIZE DESIGN

Before finalizing design, possible variations in plant processing parameters such as discharge rates or milling redesign for different grinds may have to be re-evaluated and procedures repeated starting at Step 6. The design should then be finalized subject to revisions and updates as additional data and experience is gained during operation of the pond and prior to closure.

5.16. STEP 16: OPERATION AND CLOSURE PLANS

The time available to collect initial design data may be limited, hence limiting the available data. Therefore, the data-gathering program should continue during operations in order to confirm or revise the assumptions made in selecting design events and criteria. The ongoing data collection also enhances the reliability of the database for application to the finalization of closure plans.

The operational plan should establish the schedule of enlargement for the tailings pond and the data requirements of the monitoring program. The plan should allow for new parameters to be monitored as needed and for existing programs to be phased out if their data is no longer required.

The closure plan should establish potential locations for spillways, inflows and outflows and the draining and filling requirements. The plan should establish the monitoring and analysis schedule and the reclamation procedures to be followed during closure.

Logic charts for the operations and closure phases are illustrated in Figures IV-6 and IV-7, respectively. The provide a basis for formally revisiting the initial design procedure with the benefit of additional data and/or operating experience.

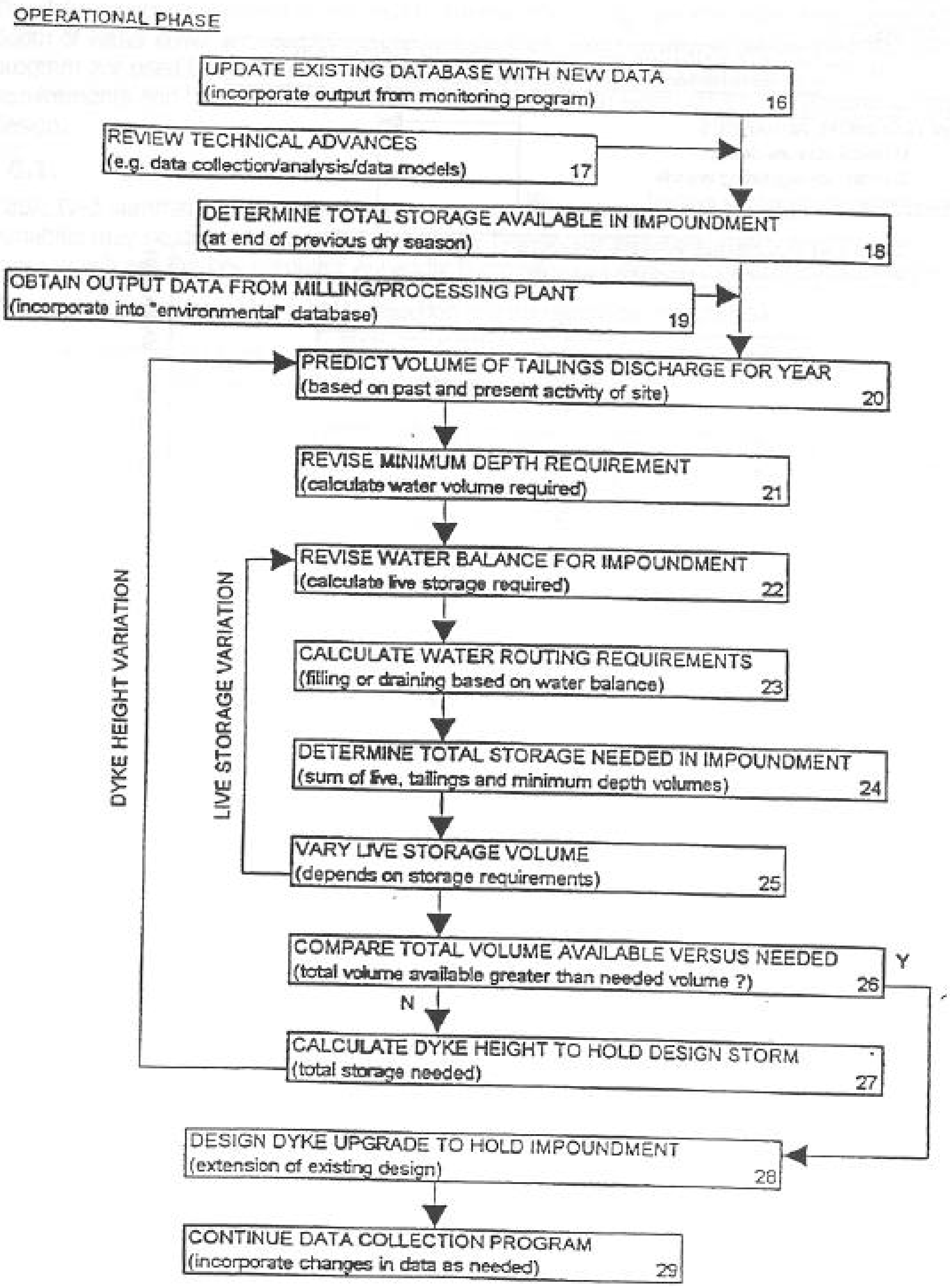


FIGURE IV-6 OPERATIONAL LOGIC CHART

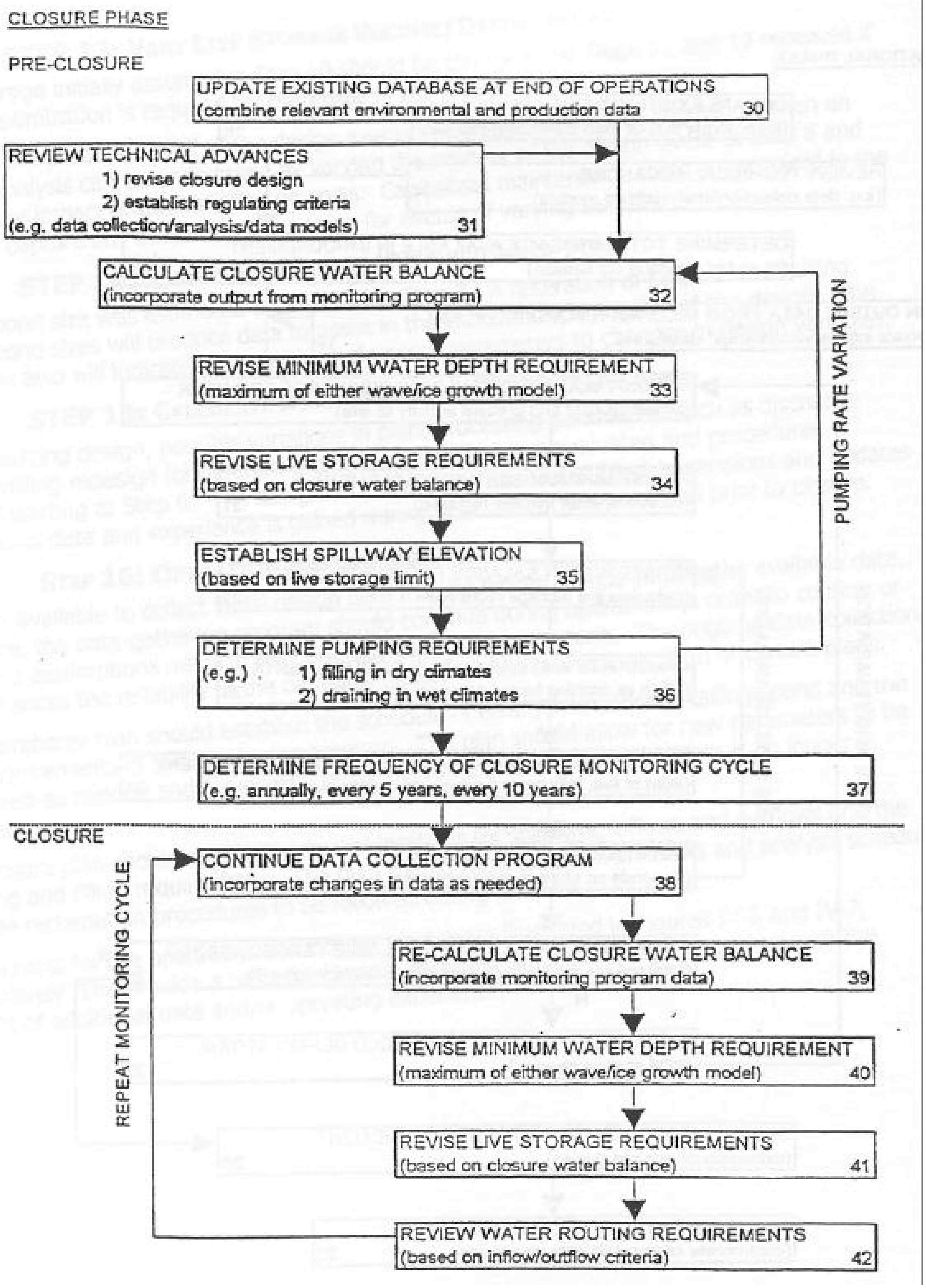


FIGURE IV-7 CLOSURE LOGIC CHART

6.0 DATA COLLECTION PROGRAMS

The data collection program is required to support the tailings pond design, and specifically the depth of water cover required to ensure bed stability. Data on key variables collected during the program are used to derive other design parameters or design requirements. The data requirements and their use are summarized below for both water cover and general tailings pond design.

6.1. WATER COVER REQUIREMENTS

Table IV-2 summarizes the data requirements for determining minimum water depth. The variables may be divided into those which may be measured, those which can be derived, and those which are the key outputs. Appendix B provides detailed descriptions of each variable.

The key variables for which a data collection program may be established include:

- wind speed, wind direction, fetch, tailings density and grain size for wave activity; and,
- water temperature, air temperature and relative humidity for ice entrainment.

These variables may be determined from data recorded in a climatological station, by measuring the geometry of the pond, and by initiating a sediment sampling program on the tailings. Particle size and density estimates of the tailings are derived from the milling design or from other similar mining operations. All remaining variables may be derived from these measurements.

6.2. WATER BALANCE REQUIREMENTS

A water balance is required for any tailing containment system. However, in a pond that requires all the tailings to be flooded a water balance is particularly critical. When dealing with flooded impoundments, dry periods and lack of water can be even more critical than floods as the tailings may dry regularly and oxidize. During dry periods, the need to maintain a minimum water cover may require that reclaim from the pond be reduced and hence new fresh water may have to be added to the system.

A schematic layout for a tailings water balance is attached identifying the components that must be addressed (Figure IV-3).

Table IV-3 summarizes the key variables required to determine water balance for pond design. Appendix B describes each of these variables.

The key variables for which a data collection program may be established include:

- precipitation, relative humidity, solar radiation, net inflow and outflow, pumped volumes, pond geometry and topographic elevations for water balance calculations.

These variables may be determined from data recorded in a climatological station, by monitoring discharges on the nearby streams, by measuring the geometry of the pond, and by surveying the topography of the site. Tailings discharges and processing flows for treatment are data inputs required from the milling and plant design. All remaining variables may be derived from these measurements.

TABLE IV-3
KEY VARIABLES AFFECTING WATER BALANCE OF POND DESIGN

Measured	Derived	Site Design	Output
Precipitation Relative humidity Solar radiation Net inflow Net outflow Topography Pond geometry	evaporation seepage surface area	tailings discharge make-up water processing needs inflow/outflow	live storage dyke height freeboard decant water

7.0 SITE SELECTION AND DAM DESIGN

In the last 30 years, the science of soil mechanics has been applied to the design and construction of tailings dams. This experience has identified several key factors that must be addressed in order to design a safe dam by today's standards (ICOLD, 1995a). These include an understanding of the following:

- seismicity and earthquakes and the corresponding response of the structures;
- hydrology and the prediction of flooding and water erosion;
- surface and sub-surface seepage of contaminants from a chemical perspective;
- seepage control with respect to minimizing excess pore pressures and preventing solid migration (piping);
- plugging of filters and drains with fines and/or precipitates; and,
- tailings dam containment systems as a whole and not just the dams in isolation.

7.1. SITE SELECTION

Site selection for a tailings containment area is generally controlled by several key factors which include:

- distance and elevation to the mill;
- topographic constraints; and,
- foundation constraints.

One important factor in selecting a site for a permanently-flooded impoundment is the quality of the foundation. The foundation condition which requires most consideration is permeability with respect to flooded impoundments. Seepage rates may not allow the maintenance of a permanently flooded impoundment.

Foundation conditions which can influence seepage include:

- high permeability soils such as sands and gravels, formed as glaciofluvial or fluvial deposits;
- residual soils with well developed joints and secondary structures that create a secondary permeability;
- Karst foundations caused by limestone dissolution can lead to seepage losses into undetected caverns; and,
- permafrost or permanently frozen ground that can thaw rapidly.

Flooded impoundments are also more susceptible than non-flooded impoundments to problems from long-term natural hazards that can allow overtopping of impoundments. Such natural hazards include:

- massive movements of earth or rock caused by gravity that slide down sloping terrain into the pond, generating a wave or filling the pond to its capacity; and,
- slides which may block spillways required for permanently flooded ponds.

Finally, when considering the long-term stability of flooded impoundments seismicity and earthquakes may require a design for a Maximum Credible Earthquake as opposed to a shorter design period for an embankment that will be drained on closure. Recent work has summarized and recommended procedures for the design of dams under Earthquake Conditions (CEA 1989)

7.2. IMPOUNDMENT LAYOUT AND DAM DESIGN

Impoundment layout must be handled simultaneously with site selection. The suitability of a site for a flooded impoundment cannot be fully established without confirming that a particular embankment design is suitable.

Impoundment designs can include Ring Dykes, Cross Valley Impoundments, Sidehill Impoundments and Valley Bottom Impoundments. Typical schematic pond layouts are shown on the attached Figure IV-8.

Advantages and disadvantages of each of the above layouts may depend upon the hydrology of the site. If the area is in a dry climate, a sidehill or valley bottom layout may not have sufficient catchment to ensure long-term flooding. Alternatively, in a wet climate, a cross valley dam may intercept too much water and this could lead to high costs for diversion, flood control, pumping, and maintenance. The risk of failure is generally increased if major diversions are needed.

There are three generic types of dams that have been used in the past. These include upstream construction, centreline construction and downstream or water retaining type construction. Typical cross sections are shown on Figure IV-9.

7.2.1. UPSTREAM CONSTRUCTION

Upstream construction is used generally when the tailings themselves are relied upon for the dam construction. The material is placed hydraulically and requires little in the way of equipment during construction. Upstream methods of construction are the cheapest method of containment. However, upstream construction relies on a coarse grind to manufacture a coarse free draining beach. Generally at least 40 to 60% of the tailing stream should be sand for this method to succeed. Upstream dams are by their nature of construction loose and saturated and are susceptible to liquefaction in areas of moderate to high seismicity. Pond placement as far as possible from the dam is the primary method of controlling the phreatic surface in the dam. Consequently, dams of this type cannot be flooded to control ARD without creating slope stability and seepage problems.

7.2.2. CENTRELINE CONSTRUCTION

A centreline method of construction is a compromise between the upstream and downstream methods in many respects. Internal drainage can be provided but spigotting and beach control is necessary so that a beach is formed for support of the small amount of fill that falls on the upstream side of the centreline. The width of the beach is not necessarily large but this type of construction precludes flooding during operations to mitigate ARD.

7.2.3. DOWNSTREAM CONSTRUCTION

The tailings are initially discharged from behind a starter dam and embankment fill is placed on the downstream slope of the starter dam. This method allows the incorporation of filters and drains for positive control of seepage and pore pressures. This type of dam can allow for storage of water against the upstream dam face making this type of dam most amenable to control of ARD by flooding.

Downstream raising requires careful advance planning as the height increases and the toe progresses outward. The major disadvantage, however, is the large volumes of embankment fills required which can financially burden a project. The availability of the large volumes of fill may be a constraint. This type of construction requires strict filter and construction control. In addition, if a beach is not built up then rip rap protection is required on the dam upstream face which is not required with a tailings beach.

Flooding for ARD control requires a downstream or at least modified downstream type of dam construction.

7.2.4. FILLING CURVES

Filling curves are an integral part of constructing tailings dams. The curves are site specific and are used to predict required dam crest elevations at any time. The curves are developed as volume-elevation curves or time-elevation curves. The curves should indicate the following:

- level of solids in the pond accounting for beaching slopes and depositional methods;
- high and low water levels in the pond accounting for wet and dry cycles and flood storage;
- freeboard requirements; and,
- crest elevations of the dykes.

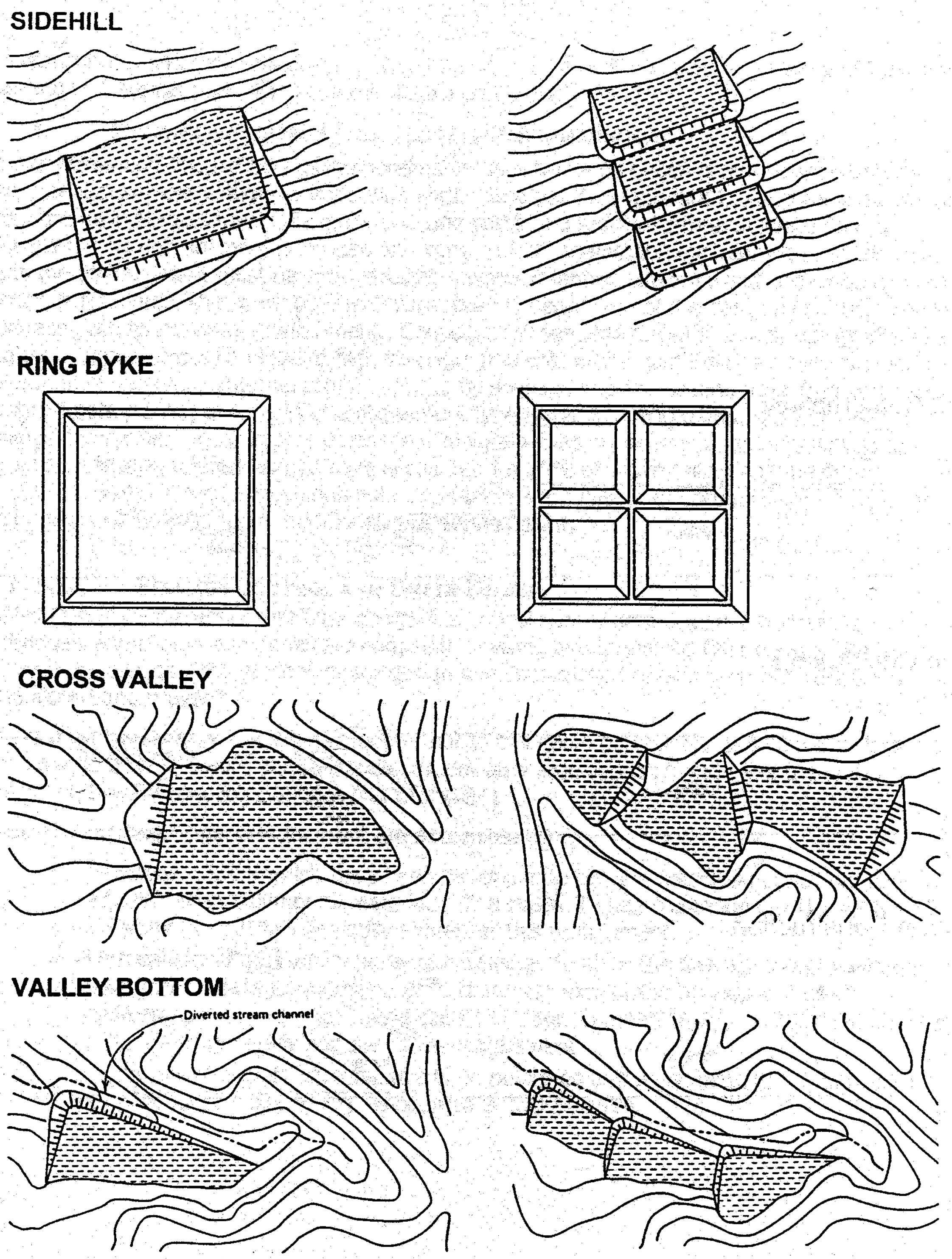
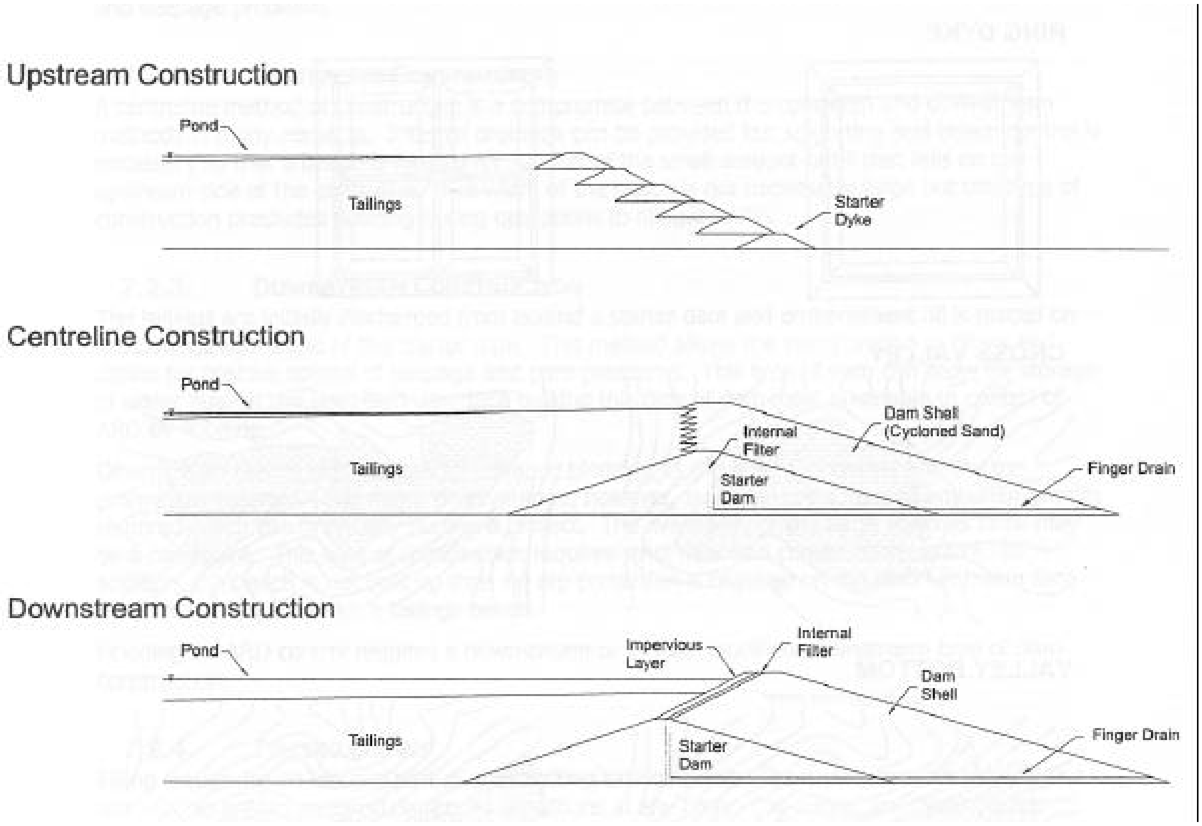


FIGURE IV-8
TAILINGS POND LAYOUTS



Upstream Construction

Centreline Construction

Downstream Construction

FIGURE IV-9
SCHEMATIC OF VARIOUS METHODS OF DAM CONSTRUCTION

Construction is generally scheduled to allow the dam to be built one year in advance of the design elevation. A typical pond filling curve is shown on Figure IV-10.

7.3. MITIGATIVE DESIGNS FOR FLOODED IMPOUNDMENTS

Conventional non-flooded tailings impoundments usually rely on beaches to at least partly control the phreatic surfaces within the impoundments. Long beaches separate the accumulated water from the main embankments, increase seepage paths and reduce internal seepage forces. However, if an embankment is flooded and water is held against the upstream face of the dam then the embankment must be considered as a water retention structure and conventional dam design is required. The water ponded against the upstream face of the dam has a much smaller seepage path to the outer environment. Consequently seepage forces and seepage gradients are higher. Higher forces can lead to high seepage gradients in the dam interior which can lead to migration of particles. Internal erosion caused by seepage was the second most frequent cause (approximately 16%) of failures of embankment type water retaining dams. (ICOLD 1996). For tailings dams where most impoundments are maintained as dry or impoundments with long beaches, seepage related failures have accounted for 10% of active tailing impoundments (USCOLD,1994). Flooded impoundments could therefore increase the probability of a seepage type failure which will require additional care during design.

7.3.1. SEEPAGE CONTROL AND DRAIN DESIGN

Ripley (1988) has stressed that filter design is a critical part of dam design. Numerous individuals and design authorities are constantly revising and improving filter design. Ripley however notes that "the technology for design and construction of piping-proof filters is long established and proven".

Filters must be designed : not to segregate; not to change in gradation; not to have apparent or real cohesion; be internally stable; have sufficient discharge capacity; and have the ability to control and seal a concentrated leak (ICOLD, 1994).

Typical filter design criteria include but are not limited to:

- Retention criteria which have been developed to prevent migration of soil particles from adjacent foundation or fill materials. The classic Terzaghi criterion $D_{15}/d_{85} <4$ (or <5 US Army COE, 1986) generally addresses this requirement.
- Permeability criteria which have been selected to allow the filter to accept seepage flows from adjacent foundation or fill materials without the buildup of excess hydrostatic pressure. Again, the classic Terzaghi criterion $D_{15}/d_{15} > 4$ (or >5, US Army COE, 1986) generally addresses this requirement.
- Numerous other refinements have been published and a complete list of state-of-the-art practices for filter design is outlined in ICOLD, 1994.

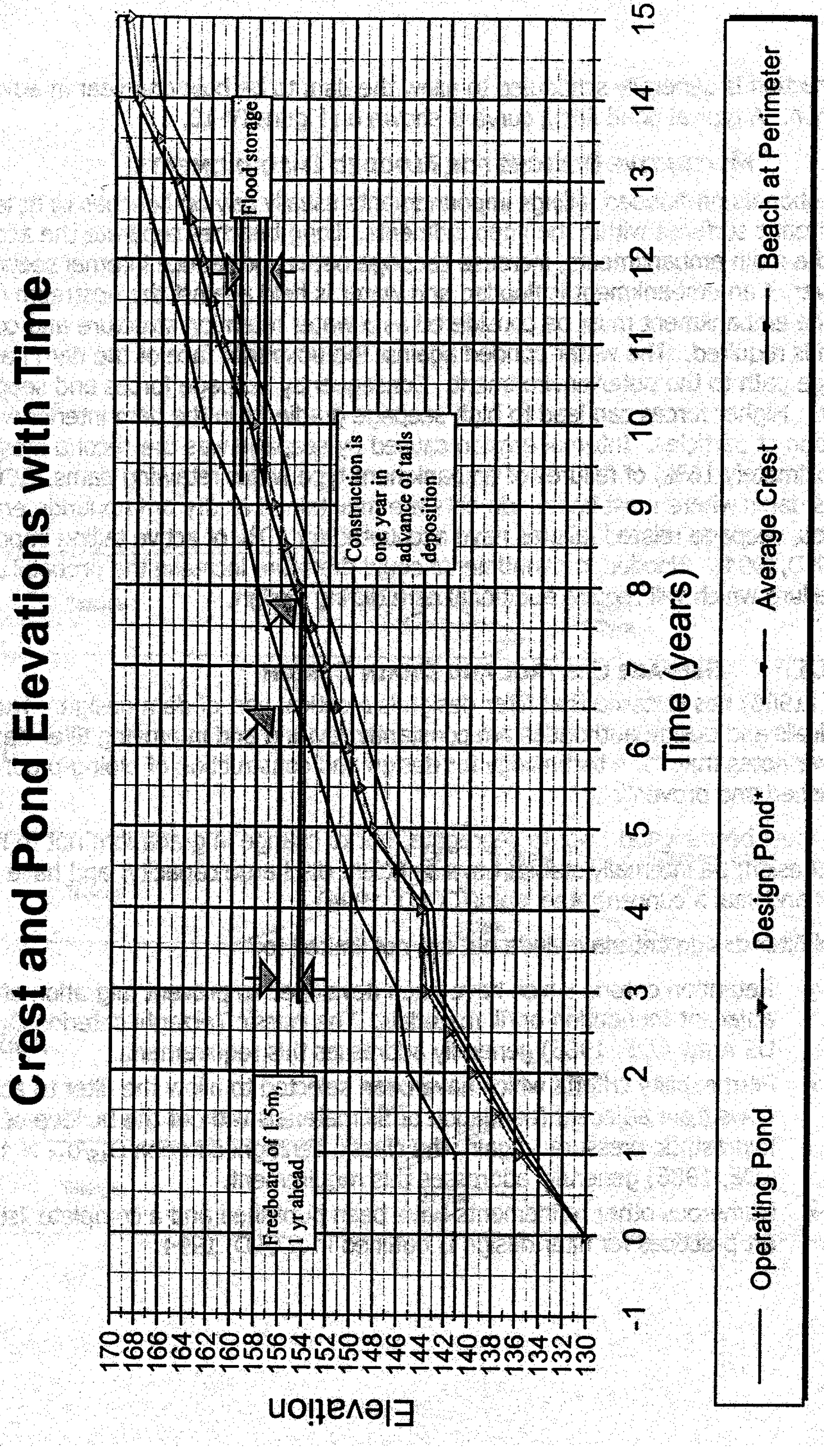


FIGURE IV-10
POND FILLING CURVE

8.0 DEPOSITIONAL STRATEGIES

In order to keep tailings from oxidizing there is a minimum depth of water cover required. If the water cover is too thin then tailings may be re-entrained into the supernatant by wave action. In order to keep tailings flooded without penalizing the height of the embankment, it is critical that the tailings surface stays level. Tailings placement and deposition plans can be a concern if the impoundment is wide and shallow. It may be necessary to resort to floating discharge lines or rock groynes built into the pond to distribute the tailings.

A Hudson Bay Mining and Smelting site in northern Manitoba, a floating pipeline was used to carry the tailings to the discharge point. The end of the pipe was controlled by ropes and manoeuverable barges. Initially, the pipeline is swung in an arc until a semi-circle is filled in. The pipe was then shortened and the process repeated. In a deep lake or deep part of a valley, it may be possible to swing the pipe every 6 to 8 weeks. In areas where a shallow lake or pond has to be filled, moves may have to be on the order of every one to three days (Wayne Fraser, private communication).

Winter operations are usually simplified by laying the pipe out on the ice over the deepest part of the impoundment. The tailings are then discharged through the ice and the line shortened as the bottom of the lake or pond fills in.

Large flat ponds may be difficult to operate using this method and may require anchors and barges.

9.0 RISK ASSESSMENT

Recent releases of tailings effluents and solids from containment facilities around the world, including Mary Spruitt (1994), Omai (1995) and Marcopper (1996), (USCOLD 1994), have heightened awareness that risks associated with tailings containment must be fully addressed during all phases of a facility life. These phases include design, construction, operation and closure of tailings impoundments.

The failure modes for the mines can be limited to malfunctions of the tailings system that would have a physical or chemical impact on the environment. However, chemical impact on the environment by ARD is being addressed by others.

In order to fully assess the risks of a tailings containment failure the “system” is defined starting at the tailings pump box in the mill and following through to final discharge of the effluent to the environment.

9.1. GOALS AND OBJECTIVES

The primary objective of a risk assessment is to provide a working document for each mine to allow assessment of design and operating risks at each site. One of the primary goals of this study is to ascertain how the mitigative/preventative structures and/or procedures already on site affect risk.

Tasks generally undertaken are:

- Identification of potential failure modes which could lead to physical release of tailings or pond supernatant into the environment;
- A qualitative estimate or, wherever possible, a quantitative estimate of the probability of failure associated with the failure modes;
- Identification of the receptors that would be impacted by any failure and wherever possible the estimation of impact in terms of cost; and
- Assessment of the risk associated with each occurrence obtained by multiplying the probability of occurrence by the “cost”.

A summary of the process is shown graphically on Figure IV-11 to assist in undertaking and understanding the assessments.

9.2. RISK ASSESSMENT DEFINITIONS AND BASIC CONCEPTS

9.2.1. RISK ASSESSMENT JUSTIFICATION

Management of natural hazards by natural resource and transportation industries has for many decades been, and in many cases still is, mainly reactive. However, over the last few years, some industries have become proactive. There has been a renewed interest in risk analysis as a means for rationalizing decision making in current times of financial constraints and limited budgetary resources. In addition, rising public awareness of risk has led to the need for higher levels of regulatory scrutiny and for modern risk-based approaches to hazard management.

Risk management is an emerging discipline that integrates risk assessment and risk control measures. In an effort to assure consistency amongst agencies adopting risk management procedures, the Canadian Standards Association (CSA) has prepared a quality management manual which provides definitions and sets out general requirements and guidelines for risk analysis (Canadian Standards Association, 1991). Definitions of terms and notations related to risk assessment are summarized on Figure IV-12.

Fell (1994) notes that an important part of the risk assessment procedure is the evaluation of the acceptability of risk. This can be done by comparing calculated risk with socially acceptable risk, usually described in terms of the assumed probability of loss of life; cost effectiveness of risk reduction commonly in terms of cost per life saved; or a cost benefit analysis, by comparing the cost of risk reduction measures with the annual reduction in risk.

There is little in the way of published statistics for tailings impoundments so the notion of relative risks is generally used. Consequently in this study the term risk is defined as the combination (a multiplication) of the likelihood of a specified hazard being realized and the consequences (harm and/or damage).

9.2.2. RISK ASSESSMENT PROCESS

An effective risk assessment initially requires identification of hazards or potential failure modes. Many of the hazards related to tailings containment facilities are unique to the mining industry. For example, mines, and in particular tailings containment systems, are constructed over long periods of time by a changing work force and usually under changing design criteria. Tailings systems are complex and include man-made components such as dams, pipelines and ponds interacting with natural components such as slopes, seismically active faults, precipitation and runoff. In addition, previous operations can have an impact on newly built components. Furthermore tailing containment methods are process specific and therefore unique to each mine.

A comprehensive risk assessment system must account for all types of hazards and the affected components existing at a specific site or location. To be effective the risk assessment approach must be systematic, yet accommodate the varying spatial and temporal considerations of each mine site.

In order to assess as many failure modes as possible and provide a template for comparison at each mine site, a series of potential failure modes were identified based on previous historical data published by United States Committee on Large Dams, (USCOLD, 1994).

Failure modes which are attributable directly to single external causes are identified as “elemental failure modes”. Elemental failure modes cannot be subdivided further. Examples of elemental failure modes are a tailings pipe bursting as a result of:

- mechanical failure;
- a traffic impact; or

- overpressurizing due to freezing or sanding.

Several elemental failure modes could independently impact on a given element or link. Such elemental failure modes can be grouped together to create “compound failure modes”. For example, a pipeline rupturing on top of the main dam has been identified as a compound failure mode which could be the result of any one of the elemental failures noted above.

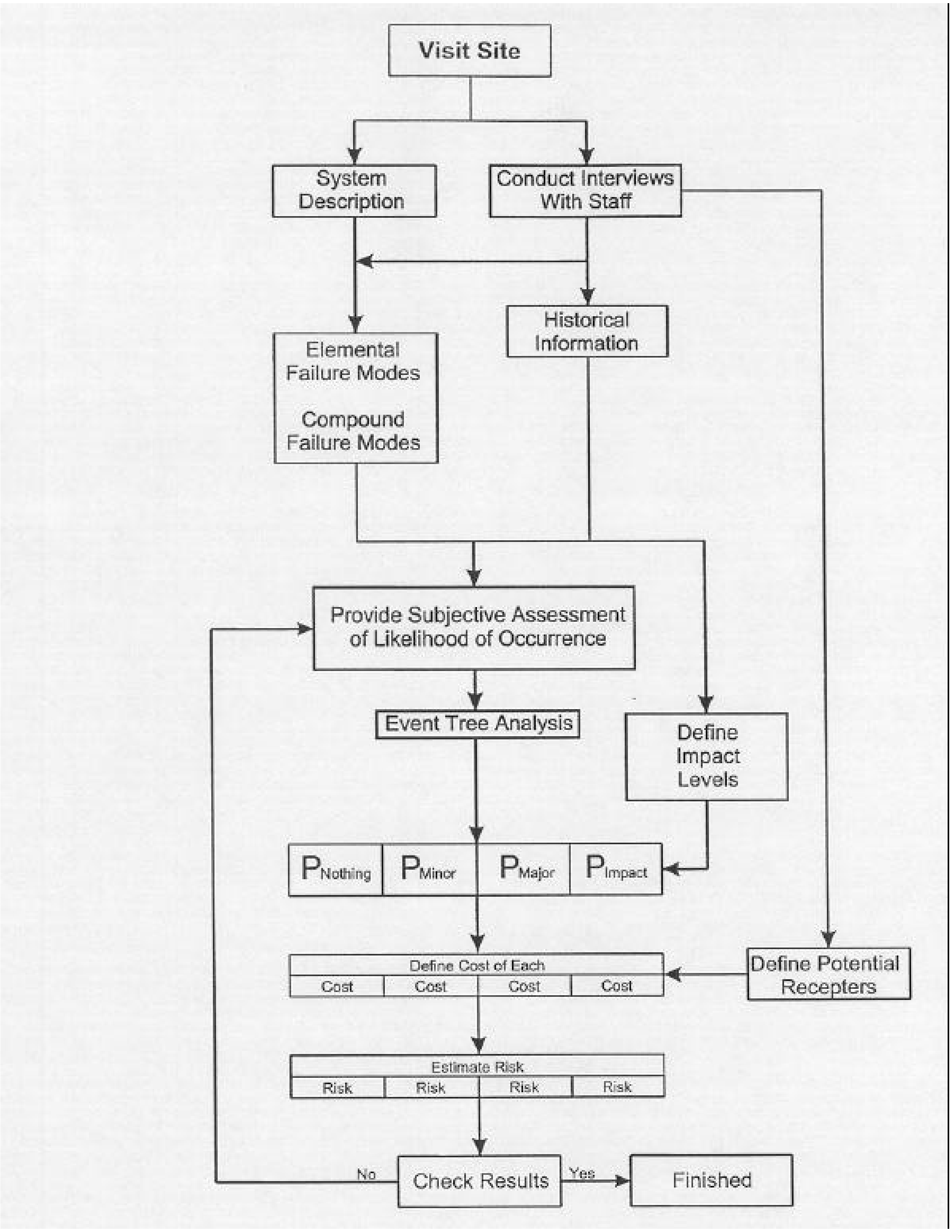
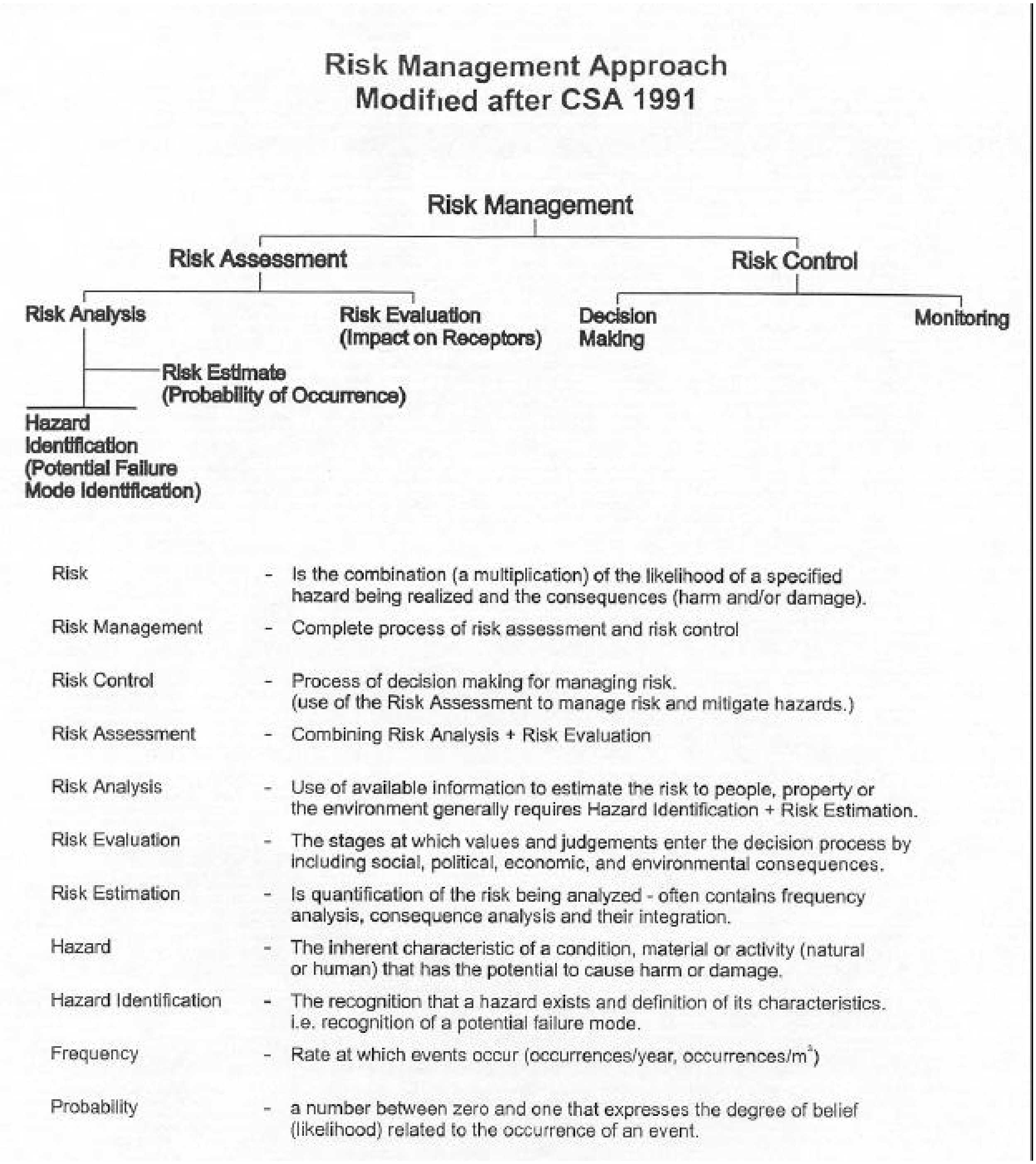


FIGURE IV-11
SUMMARY OF RISK ASSESSMENT PROCESS


Risk Management Approach
Modified after CSA 1991

Risk	-	Is the combination (a multiplication) of the likelihood of a specified hazard being realized and the consequences (harm and/or damage).
Risk Management	-	Complete process of risk assessment and risk control
Risk Control	-	Process of decision making for managing risk. (use of the Risk Assessment to manage risk and mitigate hazards.)
Risk Assessment	-	Combining Risk Analysis + Risk Evaluation
Risk Analysis	-	Use of available information to estimate the risk to people, property or the environment generally requires Hazard Identification + Risk Estimation.
Risk Evaluation	-	The stages at which values and judgements enter the decision process by including social, political, economic, and environmental consequences.
Risk Estimation	-	Is quantification of the risk being analyzed - often contains frequency analysis, consequence analysis and their integration.
Hazard	-	The inherent characteristic of a condition, material or activity (natural or human) that has the potential to cause harm or damage.
Hazard Identification	-	The recognition that a hazard exists and definition of its characteristics. i.e. recognition of a potential failure mode.
Frequency	-	Rate at which events occur (occurrences/year, occurrences/m^3)
Probability	-	a number between zero and one that expresses the degree of belief (likelihood) related to the occurrence of an event.

FIGURE IV-12
RISK MANAGEMENT APPROACH MODIFIED (CSA, 1991)

9.3. RISK ASSESSMENT ANALYTICAL METHODS

Development of a document for analysis of complex facilities and complex potential failure phenomena relevant to tailings containment systems requires four simple steps.

- Choice of risk assessment analytical methods (HAZOP, FMEA *etc.*)
- Identification of hazards or failure modes
- Estimation of the probability of occurrence
- Identification of receptors and risk evaluation

These are discussed in the following sections.

9.3.1. AVAILABLE ANALYTICAL METHODS

A variety of techniques are available for assessing “what can go wrong” and estimating its probability of occurrence. These include but are not necessarily limited to:

- Potential Problem Analysis (PPA);
- Failure Modes and Effect Analysis (FMEA);
- Hazard and Operability Study (HAZOP); or
- Event Tree Analysis (ETA).

A Potential Problem Analysis (PPA) asks two basic questions:

- What could go wrong?
- What can we do about it now?

A PPA generally includes the following activities:

- identification of areas of vulnerability;
- identification of potential problems within the areas of greatest vulnerability;
- identification of actions that will prevent the occurrence of specific potential problems; and
- identification of contingency actions which would minimize the effects of potential problems that cannot be prevented.

A Failure Modes and Events Analysis (FMEA) is a primarily qualitative technique which can be quantified to a degree. In an FMEA, the effects or consequences of individual component failure modes are systematically identified. The analysis is usually descriptive and is organized using a worksheet or table to display the information. FMEA relates component failures modes and their causative factors and effects on the system and presents them in a readable format. The major disadvantages to the FMEA are the difficulties in dealing with generalized risks and comparing these with other assessments to allow ranking of risks.

A HAZOP is a form of FMEA which was originally created for the chemical industry. It provides a rigid systematic way of identifying hazards and operating problems throughout an entire facility. The basic objectives of the technique are to:

- produce a full description of the facility or process including the intended design conditions;
- systematically review every part of the facility or process to discover whether deviations from the design have occurred; and
- consider how these deviations could lead to hazards.

The principles of HAZOP can be applied to systems under operation or in various stages of design. A HAZOP study requires:

- a team which consists of design, operating, and risk personnel;

- collection of design documents including operating and maintenance manuals;
- definition of the scope of the study;
- identification and analysis of each major component in the system and all supporting systems; and
- documentation on HAZOP worksheets of the consequences of deviations from design and a summary of those deviations which are considered hazardous and credible.

An Event Tree Analysis (ETA) is a technique, either qualitative or quantitative, that is used to identify possible outcomes and their probabilities given the occurrence of an initiating event. It is generally assumed that each event in the sequence is either a success or failure. An ETA is an inductive type of analysis in which the basic question addressed is “what happens if...?”.

An Event Tree Analysis provides a graphic means of representing potential failure scenarios and allows quantification of the probability of each failure mode. However, it was considered unlikely that sufficient data would be available to fully support an ETA assessment. Therefore aspects of the FMEA and PPA techniques as well as the ETA are often incorporated to allow consideration of failure modes with semi-quantitative data.

9.4. HAZARD OR FAILURE MODE IDENTIFICATION

9.4.1. ELEMENTAL FAILURE MODES

A preliminary list of elemental failure modes for tailings dams has been developed by reviewing a report on tailings dam incidents and failures compiled by the United States Congress on Large Dams (USCOLD,1994) and an update on tailings dam incidents from 1980-1996 prepared by the United Nations Environment Programme (UNEP, 1996).

The reports indicated that tailings dam failures fell into the following categories:

- dam overtopping;
- slope instability;
- earthquakes;
- foundation;
- seepage;
- structural;
- erosion;
- mine subsidence; and
- unknown.

These were considered by USCOLD to be potential failure modes that required assessment in a Risk Analysis Program.

The summary in Table IV-4 shows that slope instability is the leading cause of both failures and accidents in *active* tailings dams. This is closely followed by earthquakes. Overtopping, foundation and seepage have all occurred with similar frequencies of failures and accidents, while erosion and mine subsidence account for only a minor proportion. However, overtopping is the principal cause of failures of *inactive* impoundments.

A more detailed set of potential failure modes are summarized in Table IV-5. The failure modes identified are all considered to be simple events and hence have been classified as elemental failure modes. Each elemental mode could act independently or in conjunction with others to lead to a tailing dam incident.

9.4.2. COMPOUND FAILURE MODES

Seven key compound failure modes numbered C1 through C7 have been defined. These are summarized below and in Table IV-6.

C1) ponding on the berms or dykes caused by excessive precipitation or unusual meteorological conditions which could lead to problems of erosion, stability and sloughing (e,s,s);

C2) a bursting of distribution line on the dam, allowing either direct escape of slurry to the environment or causing damage to the dam leading to a dam stability issue, (e,s,s);

C3) dam overtopping caused by a variety of potential failure modes which could lead to e,s,s;

C4) dam slope instability caused by a variety of failure modes including seepage problems, seismic liquefaction, static liquefaction *etc.*;

C5) rupture of the main tailings line between the mill and the dam leading to a loss of slurry to the environment;

C6) failure of the sediment or polishing pond downstream of the tailings dam leading to a release of effluent; and

C7) a spill at the mill which escapes the emergency containment pond and releases slurry or effluent to the environment.

9.4.3. FLOODED IMPOUNDMENTS

In order to mitigate the onset of oxidation, tailings impoundments have been proposed and/or built with the entire impoundment surface flooded and water ponded against the upstream face of the dam. The compound failure modes C3 and C4, overtopping and piping issues are directly relevant to this type of impoundment.

Water directly against the dams means that the seepage gradients are higher and there is less protection against overtopping as compared to dams where the ponds are kept a large distance from the upstream dam face. A flooded impoundment must therefore be built with the more conservative design parameters required for water retaining structures.

TABLE IV-4
FAILURE AND ACCIDENT CAUSES FOR TAILINGS DAMS

	Active Dams (still receiving tailings)			**Inactive Dams (no long receiving tailings)**		
Cause	**Failures**	**Accidents**	**Total**	**Failures**	**Accidents**	**Total**
Overtopping	13	3	16	4	0	4
Slope Instability	22	18	40	1	1	2
Earthquake	18	5	23	0	10	10
Foundation	8	10	18	1	0	1
Seepage	10	10	20	0	0	0
Structural	7	6	13	0	1	1
Erosion	0	2	2	0	0	0
Mine Subsidence	3	0	3	0	0	0
Unknown	16	0	16	3	0	3
Totals	**97**	**54**	**151**	**9**	**12**	**21**

TABLE IV-5
ELEMENTAL FAILURE MODES FOR IMPOUNDMENT SYSTEMS

Reservoir (overtopping)

1 Landslide into reservoir generates a wave which overtops the dam
2 Wave action overtops dam
3 Perimeter bypass system fails and water enters reservoir exceeding capacity of spillway or storage or an external creek diversion failed and water entered reservoir
4 Pond allowed to reach crest of dam due to poor operations
5 Pond allowed to reach dam by design (discharge from top end of pond to save dam height)
6 Excessive precipitation exceeds storage capacity
7 Water balance not maintained (human error)

Dam (upstream or downstream instability)
8 Seepage causes piping and removes dam material (*ie* filter fails)
9 Seepage raises pore pressures and causes shallow instability
10 Seepage raises pore pressures and causes deep instability
11 Seismic liquefaction of dams
12 Seismic deformation of dams
13 Seismic liquefaction of tailings leads to erosion
14 Liquefaction of tailings applies horizontal thrust to dam
15 Non seismic liquefaction of dam due to straining or increased pore pressures
16 Seepage failure raises pore pressures and triggers a slide
17 Construction pore pressures rise and slope moves
18 Saturation of uncompacted fill either by first fill or rain or snow encapsulated in dam fill melts, dam settles, overtops
19 Uncontrolled toe erosion retrogresses
20 Dam face erodes due to uncontrolled precipitation or snow melt

Foundation
21 Karst collapses beneath dam
22 Collapse due to mine subsidence allows tailings to escape into mine or void
23 Sliding on weak soil or liner interface
24 Compression of weak soils leads to cracking of dam
25 Permafrost degrades
26 Construction pore pressures rise and foundations move
27 Seepage through a poor membrane or pervious soils into groundwater system, bypassing seepage recovery systems
28 Seismic liquefaction of foundations
29 Seismic deformation of foundations
30 Non seismic liquefaction of foundations

Structural
31 Piping around a culvert or decant pipe
32 Reclaim tower fails
33 Decant plugs
34 Pumps fail due to loss of power
35 Conduit fails
36 Landslide blocked spillway
37 Ice blocks spillway

TABLE IV-6
COMPOUND FAILURE MODES AND THEIR CAUSATIVE ELEMENTAL FAILURE MODES

Compound Failure Modes		Elemental Failure Modes
C1	Ponding on Berm	Unfavourable Meteorologic Conditions, Geometric Susceptibility to ponding
C2	Distribution Line Bursts on Dam	Joint Breaks, Line Sands up, Line Freezes, Traffic Accident, Snow Removal Accident, Sabotage, Mechanical Rupture
C3	Dam Overtops	Slide Into Reservoir, Perimeter Ditch or Diversion Fails, Design Precipitation is Exceeded, Tails Liquefy and overtop dam, Differential Settlement of Dam, Reclaim System Fails, Sabotage
C4	Slope Instability	Seismic Liquefaction of Dam or Foundation, Static Liquefaction of dam or Foundation, Seepage Pore Pressure rise or piping, Mechanical Failure(culverts etc), Geological Anomaly (*e.g.*, karst, permafrost) leads to failure, External Physical Erosion, Sabotage.
C5	Main Line Bursts and Tailing Slurry Escapes	Slide Onto Line, Ditch Diversion Fails, Slope Liquefies, Joint Breaks, Line Sands up, Line Freezes, Traffic Accident, Snow Removal Accident, Sabotage, Mechanical Rupture
C6	Sediment Pond/Polishing Pond Fails and Overtops or Non-Compliant Water is Released.	Generalized but Site Specific Problems see Method 6
C7	Spill at Mill Escapes Containment and Slurry Reports to the Environment.	Generalized but Site Specific Problems see Method 7

V - OPERATIONAL MONITORING OF TAILINGS POND

1.0 INTRODUCTION

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

- verify initial design assumptions (*e.g.*, water balance, pond water quality, mass loadings);
- update closure design;
- fine tune operations and optimize operating costs;
- trouble-shoot water quality issues which may arise; and
- satisfy regulatory requirements.

In addition to developing confidence with stakeholders, establishing sound monitoring practices will also limit the liability of corporate management.

The strategies outlined in the following sections represent the types of monitoring that can be easily conducted by in-house mining staff. The only exception to this are those more specialized diagnostic techniques that may be required to resolve specific contamination issues.

2.0 TAILINGS CHARACTERISTICS

In order to characterize accurately all tailings-related inputs to the pond, regular monitoring of the mill discharge is essential. This will include various analyses of the liquid and solid fractions of the tailings slurry. The following section has been divided into: ARD verification, mineralogy and slurry supernatant. Each will be described.

2.1. ARD VERIFICATION

The acid generating potential of tailings should be assessed periodically during operations. Testing of tailings ARD characteristics should coincide with known changes in ore type or geology, otherwise, duplicate samples should be collected from the tailings stream once every three months.

Based on determinations of net neutralization potential, tailings should be classified as potentially acid generating or non-acid generating. These results will contribute to an ARD database for the discharged tailings and will be used in support of the final closure design. If the ARD data consistently indicate non-stat acid-generating tailings, and that an adequate thickness of neutral tailings exists over potentially acid-generating tailings, one final closure option may be to develop a drained impoundment. The benefits of such a scenario would be reduced risk of dam failure, reduced long-term monitoring costs and a potential reduction in closure commitments (*e.g.*, bonding requirements). However, detailed engineering would be needed to address all potential concerns.

2.2. TAILINGS SOLIDS CHARACTERIZATION

Knowledge of the mineralogic history of the tailings discharged to the pond during operations will ease identification of potential contaminant sources, should pond water quality deteriorate. Mineralogical examination of the tailings solids should be conducted if the ore characteristics, metallurgical processes or reagents used in the mill change significantly. These assessments should identify major sulphide minerals and other primary and secondary minerals whose solubility could affect pond water quality. Although routine mineralogical assessments are less essential if the ore type or mill processes do not change, verification monitoring should be conducted periodically.

The geochemical data typically gathered as part of the engineering feasibility and background characterization should be consulted to establish a relevant list of metal parameters for the tailings solids. Based on the operational database, the list of metals warranting ongoing analysis should then be reduced to target specific elements.

2.3. SUPERNATANT

Mass loadings associated with the tailings supernatant represent the largest potential impact to pond water quality. Accordingly, the composition and rate of input of the supernatant requires continual characterization during operations. In addition to verifying the initial design criteria, such monitoring can be used to forecast changes in pond water quality. This has obvious implications with respect to regulatory criteria, water management and treatment strategies.

During the initial mill start-up period, assessments of the flow rate and composition of the slurry supernatant should be conducted weekly in order to establish the short-term variability of the input and to flag potentially problematic contaminants. The database compiled from various sources should be used to develop an initial list of parameters to be measured. The list should be supplemented with the metallurgical information defined above which may identify problematic process chemicals and/or by-products. Any anomalous parameters identified in the pre-production phase should be added. A list of possible parameters is presented in Table V-1; however, every operation will have a site-specific list. Once the variability of the mass loadings has been assessed, the sampling frequency may be reduced (*e.g.*, monthly) and the analyses tailored to measure significant parameters only. However, the program should include more rigorous analyses once every six months (for verification of the previous assumptions) or whenever the ore or mill process changes. All data collected should be integrated into the physical and chemical database described previously.

TABLE V-1
TYPICAL PHYSICAL PARAMETERS IN TAILINGS SUPERNATANT

<table>
<tr><th>Physical Parameters</th><th>Total Metal Scans</th></tr>
<tr><td>pH
dissolved oxygen
total suspended solids
conductivity
flow rate</td><td rowspan="7">arsenic
antimony
cadmium
chromium
cobalt
copper
iron
lead
manganese
mercury
molybdenum
nickel
selenium
tin
zinc</td></tr>
<tr><th>Nutrients</th></tr>
<tr><td>Nitrate
phosphate
ammonium</td></tr>
<tr><th>Sulphate</th></tr>
<tr><th>Cyanide</th></tr>
<tr><td>total cyanide
WAD cyanide
thiocanate</td></tr>
</table>

Note: In order to assess the composition of the slurry supernatant, the liquid or dissolved fraction must be separated from the bulk tailings. The list of metals to be analyzed will be site specific, and may not include all of those specified above. Similarly, cyanide species need be analyzed only at operations where cyanide is used in ore processing.

3.0 OTHER FLOWS TO POND

“Other flows” to the tailings pond may include waste rock seepage, mine-site runoff, natural runoff, mine/pit water, treated effluents, precipitation, storm water. In order to verify initial design assumptions, regular monitoring of such inputs is essential. This will also include monitoring of other inputs which were not predicted during the design phase but which may develop during operations.

The monitoring strategy suggested for the tailings discharge should be adopted for the monitoring of other inputs to the tailings pond. This will involve more frequent sampling during the early operational stages (weekly or monthly) where a wide range of parameters are measured, followed by a less rigorous program once the variability associated with such flows has been assessed. Flow rates of the various inputs must also be measured in order to establish the mass loadings of various contaminants to the pond (Appendix D), as well as to permit better characterization of the pond water balance. The need for higher resolution sampling should respond to variability associated with local climate conditions (*e.g.*, spring freshet, precipitation). Monitoring strategies specific to the various inputs mentioned above are discussed below.

3.1. SEEPAGE FROM WASTE ROCK

Monitoring of the flow of acid drainage from waste rock to the tailings pond (*i.e.*, flow rates and composition) should be conducted frequently in the early operational stages to assess the short-term and/or seasonal variability. Regular monthly or bi-monthly monitoring should be required thereafter (Table V-2). Since inputs from waste rock sources will depend on variations in precipitation and snow melt, monitoring should be conducted during both high runoff and low-flow periods. Changes in waste rock composition and waste rock placement may also warrant the need for additional sampling.

3.2. MINESITE RUNOFF

Mass loadings to the tailings pond from mine-site runoff can usually be estimated by using precipitation measurements (to get flow rates) and by measuring water quality at one or two input points. Such calculations usually have more relevance to water management than to contaminant inputs. Variable loadings in response to seasonal variability (e.i., changes in precipitation, snow melt) should be assessed with the appropriate sampling frequency (Table V-2).

3.3. NATURAL RUNOFF

Runoff from natural drainages is usually expected to contribute little to the contaminant burden in a tailings pond but will influence water quality via dilution effects. Baseline water quality information should provide a solid database to characterize these inputs in the event that flows cannot be routed around the tailings pond. In general, natural runoff inflows should be limited in order to optimize the water balance in wet climates. In arid areas, these flows may be critical in obtaining a water cover.

TABLE V-2
PROPOSED SAMPLING FREQUENCY FOR VARIOUS TAILINGS POND INPUTS AND THE VARIABLES WHICH MAY DICTATE CHANGES TO MONITORING STRATEGIES

Input	Initial Sampling[1] Frequency	Regular Sampling Frequency	Variables
waste rock seepage	weekly	monthly	precipitation runoff, waste rock composition, waste rock placement, duration of exposure
minesite runoff	monthly	quarterly	precipitation, runoff
natural runoff	baseline data	annually	precipitation, runoff
mine/pit water	weekly	monthly	precipitation, runoff, seepage characteristics of mine/pit, pumping schedule
treated effluents	weekly	monthly	composition of untreated inflow, ore composition, discharge rate

[1] Sampling entails measurements of flow rate and chemical composition.

3.4. MINE/PIT WATER

Acidic, metal-laden inputs from pumped mine/pit water represent a potentially large impact on the water quality of the tailings pond. Accurate estimates of the mass loadings and the variability associated with these inputs should be determined in the early operational stages (Table V-2). In addition to the seepage characteristics of the mine/pit, variations in precipitation and runoff can be expected to affect the required discharge rate from these sources. Sampling should therefore be conducted accordingly to assess short-term and/or seasonal variability (Table V-2).

3.5. TREATED EFFLUENTS

Treated effluents may introduce potentially reactive solid phases (*e.g.*, ARD treatment sludges) to ponds. Characterization of treatment products (discharge rates and composition) should be conducted immediately following the commencement of discharge. This should include the water quality of the effluent supernatant (Table V-1) and mineralogy of the associated solid phases. Once the variability of these inputs has been assessed, monitoring can be conducted less frequently. However, additional sampling should be conducted immediately following anticipated changes in flow or composition of the discharged effluent, which could be induced by, for example, ore mineralogy and/or the composition or flow rate of the tailing pond inflows (Table V-2).

4.0 WATER /SOLID BALANCE IN POND

The following data collection program is required to support the tailings pond design, and specifically, the depth of water cover required to ensure bed stability. Data on key variables collected during the program are used to derive other design parameters or design requirements. The link between the data requirement and the use made of the data is summarized below for both water cover and general tailings pond design.

4.1. WATER COVER REQUIREMENTS

In order to minimize sediment resuspension in the tailings pond due to the actions of wind and ice, minimum pond depth criteria are established during the design phase. After the mine has begun operating, the physical variables used during the pond design must be compared with actual field measurements. The task of monitoring critical depth is therefore two-fold:

- to verify that the key physical variables that were used in critical depth calculations are similar to the operational scenario; and
- to ensure the critical depth is being satisfied throughout the tailings pond.

The physical variables affecting minimum depth criteria are based on conservative estimates, and as a result, are likely to reflect appropriate operating conditions. Ongoing verification of these parameters (*e.g.*, wind speed, fetch, grain size) is usually not required. However, should there be any significant deviations from the initial design assumptions, the critical depth may require review.

The variables required for the calculation of a minimum water cover may be divided into those which may be measured, those which can be derived and those which are the key outputs. The key variables for which a data collection program may be established include: wind speed, wind direction, fetch, tailings density and grain size for wave activity; and, water temperature, air temperature and relative humidity for ice entrainment (Table V-3).

TABLE V-3
VARIABLES AFFECTING MINIMUM DEPTH CRITERIA

Process	Measured	Derived	Output
wave activity	wind speed wind direction fetch tailings grain particle size tailings density	wind duration wave period wave height	bed velocity minimum depth
ice entrainment	water temperature air temperature relative humidity	ice duration days below freezing	ice thickness minimum depth

TABLE V-4
VARIABLES AFFECTING WATER BALANCE FOR POND DESIGN

Measured	Derived	Site Design	Output
precipitation relative humidity solar radiation net inflow net outflow topography pond geometry	evaporation seepage surface area	tailings discharge make-up water processing needs inflow/outflow decant water	live storage dyke height freeboard

These variables may be determined in a climatological station installed to record the necessary data, by measuring the geometry of the pond, and by initiating a sediment sampling program on the tailings. Size and density estimates of the tailings are derived from the milling design or from other similar mining operations. All remaining variables may be derived from these measurements (Table V-4).

4.2. WATER BALANCE

The key variables for which a data collection program may be established for water balance calculations, are illustrated in Table V-4 and include:

- precipitation, relative humidity, solar radiation
- net inflow and outflow, pumped volumes; and
- pond geometry and topographic elevations.

The first set of variables may be determined by the installation of a climatological station . Net inflow might include consideration of minewater, non-diverted runoff, drainage from waste rock dumps, groundwater inflow and recycled seepage. Net outflow might include supernatant decant, non-recycled seepage and groundwater ouflow. Measurements to estimated these flows might include monitoring discharges on nearby streams, and directly measuring each individual flow. Tailings discharges and processing flows are data inputs which can be obtained from the milling and plant design. Specific data will include: production rate for solids, slurry density, reclaim water flows, and make-up water requirements. Measurement of the geometry of the pond by site surveys is also required to develop the tailings pond volume-elevation curves. All remaining variables may be derived from these measurements.

An initial comparison of actual operating conditions with the design variables is needed after operations commence. It is important to ensure that the critical water depth is maintained throughout the pond. This will require annual bathymetric surveys using a sounder or plum line. The need for more rigorous monitoring should respond to modifications in depositional practice and changes in the water balance (Table V-3). However, the maintenance of the pond water balance and filling schedule should obviate the need for frequent surveys.

5.0 POND CHARACTERISTICS

Operational monitoring of a tailings pond entails the development of sampling strategies directed towards various components of the pond system including: pond water quality; sediment and pore-water chemistry; and groundwater. The following sections provide guidelines for the operational monitoring of a subaqueous tailings disposal facility; however, strategies should be tailored to address site-specific characteristics of a mining operation.

5.1. WATER QUALITY

Direct monitoring of the pond supernatant provides immediate evidence for relevant changes in the impoundment. More specifically, continual monitoring can be used to:

- verify design assumptions for operational water quality;
- predict contaminant loadings to the receiving environment;
- ensure that operational pond water quality meets regulatory criteria (*e.g.*, for cyanide);
- identify contamination problems associated with the post-depositional reactivity of tailings solids and other potential problems not predicted in the design phase; and
- verify treatment requirements for discharge to the pond and/or discharge to the receiving environment.

Tailings pond water quality should be assessed frequently during the early operational stages. Weekly monitoring for a wide range of parameters is recommended for the first few months of operation or until the pond has approached a steady state (Table V-1). Once the levels of the various parameters have come into steady-state, sampling should be conducted less frequently (*e.g.*, monthly) while focusing on specific parameters of concern. The time period for a particular pond to approach steady-state will depend on several variables including the pond volume, pond residence time and contaminant loading.

5.2. SEDIMENT/PORE-WATER GEOCHEMISTRY

Sampling of the sediments and associated pore-waters in the tailings pond is not recommended for routine operational monitoring. Such sampling, which can be used to assess the post-depositional reactivity of submerged tailings, should be used as a diagnostic tool to assess the release of contaminants from the tailings to the pond supernatant or groundwater should water quality become an issue (see Table V-5). The application of such monitoring is described below.

When a water quality problem develops in the tailings pond, several steps can be taken to identify the source, or sources, of the contamination. First, examination of the predicted mass loadings from the chemical and physical database may point to problematic inputs. However, a comparison of the mass loadings into and out of the impoundment may not always identify the contaminant source. The mass balance calculations described above, for example, do not take into account loadings to the pond supernatant from reacting tailings solids. In order to assess such reactivity, pore-water sampling techniques can be applied. These methods (*i.e.*, dialysis arrays and coring/centrifugation) can be used to assess whether contaminants are being released from the tailings to the overlying pond waters. Pore-waters should be collected in an area of the pond at which tailings are not rapidly accumulating. This will permit the sampling of "aged" tailings solids which have approached near steady-state equilibrium. Knowledge of the depositional history of the pond (*i.e.*, timing and placement of various solid phases) will influence the choice of sampling locations.

The strategy described above may also apply to situations where groundwater contamination is a concern. An assessment of the post-depositional reactivity of the tailings solids can indicate whether the post-depositional release of contaminants is responsible for metal inputs to groundwater.

5.3. GROUNDWATER

The construction and operation of a water-retaining tailings impoundment generally results in an artificial mounding of the water table beneath the structure. Tailings impoundments are therefore often areas of groundwater recharge where downward hydraulic gradients result in process water migrating into the underlying natural groundwater system. The monitoring of groundwater in the vicinity of the tailings impoundment is necessary to provide both an early seepage detection system and to generate important hydrogeological information (*e.g.*, hydraulic head, gradients, hydraulic conductivity).

A number of questions need to be addressed in an operational and post-closure groundwater monitoring program:

- What is the predominant direction and rate (velocity) of regional groundwater flow in the area?
- What is the predominant direction and rate of groundwater flow from the tailings impoundment?
- How do groundwater flow rates vary seasonally from the impoundment?
- What is the chemistry of groundwater up-gradient of the tailings impoundment and down-gradient of the impoundment?

Monitoring wells should be installed upgradient of the impoundment (control), around the perimeter and immediately downgradient. In order to determine hydraulic gradients, a second set of monitoring wells at a distance from the original wells should also be installed. The number of monitoring wells required will depend on a number of site specific conditions (topography, geology, impoundment geometry, *etc.*).

5.3.1. PHYSICAL HYDROGEOLOGY

The tailings pond design will normally include an assessment of the groundwater system around the pond and potential seepage scenarios. These defined flow paths should be used to develop a network of monitoring wells. It is beyond the scope of this design guide to detail the specific details of hydrogeology assessments. However it is important to emphasize that there is a need to effectively manage this area and hence an engineered system of groundwater wells and an associated monitoring program is recommended.

The monitoring frequency of physical hydrogeological parameters will vary. Water level determinations (hydraulic head) are straightforward and inexpensive and can be conducted by mine technical personnel. This monitoring should be conducted at most monthly when groundwater recharge is greatest (freshet), and quarterly because of the generally slow rates of movement in groundwater systems.

5.3.2. CHEMICAL HYDROGEOLOGY

The primary objective of a groundwater chemistry monitoring program is to detect any alterations to backgroundwater chemistry that would be indicative of significant seepage from the impoundment. Combined with physical hydrogeological information, impacts to groundwater and surface water receptors can be assessed and mitigation measures proposed.

Since flow and response rates are generally much slower in groundwater systems, quarterly sampling would be adequate in the absence of any material/process changes. Should changes to groundwater chemistry exceed fluctuations observed during baseline monitoring prior to operations, more frequent (monthly) sampling should be conducted.

These general descriptions outline the fundamental elements and key questions of a groundwater monitoring program. The implementation of the optimum system for a specific site requires engaging the services of a hydrogeologist. Similarly, the interpretation of the acquired groundwater data requires combining proper hydrogeological expertise with groundwater flow and geochemistry.

6.0 POND DISCHARGES

One of the assumptions made in this design guide is that there will be no control of pond water quality during operations. In addition to high concentrations of suspended solids, the pond may host high levels of trace metals. If discharge to the receiving environment is required, pond effluents will likely require some form of treatment to reduce contaminant levels and to satisfy specified regulatory criteria.

Monitoring of the pond during operations will yield data which can be compared to the original design assumptions. In addition, any significant changes in operations may be reflected in the tailings pond and hence a routine sampling program is needed for anticipating these changes and allowing adjustment of the treatment plant operations. Such sampling frequency may range from weekly to monthly depending on many variables such as the size of the pond and seasonal variability due to pond freezing or water column stratification. The parameters of interest will have been determined in the design phase and will include specific metals, general chemical parameters and suspended solids.

The discharge from a tailings pond will normally be governed by a permit to discharge into receiving waters. A monitoring program for both the discharge water and the receiving environment will be associated with such an operation. As stated previously, it is unlikely that direct discharge will occur without some form of treatment to remove suspended solids and/or trace metals.

7.0 SUMMARY

A summary of the recommended sampling frequencies for the various pond characteristics is presented in Table V-5. Considerations which may warrant modifications to monitoring strategies are also presented. It should be re-emphasized that due to the diversity of various mining operations, monitoring programs should be tailored as required to meet site-specific requirements.

8.0 GEOTECHNICAL MONITORING

The objective of geotechnical monitoring programs is to routinely:

- assess the stability of structures (*e.g.* dykes);
- validate design elements; and
- make any changes where needed as indicated by field observations.

The programs should be under the direction of a geotechnical consulting firm. The designs of such programs are dependent on each geotechnical design and on the results of the risk assessments conducted to highlight key areas for ongoing surveillance. In general, these programs are intended to confirm that the geotechnical designs are sound and that structures are performing as specified. Most of the routine monitoring can be performed by trained technical staff, with overviews conducted by experts on an annual or semi-annual basis.

The scope and frequency of the monitoring may be defined by a regulatory permit. In addition, programs may be governed by corporate policies established by individual companies to minimize risk and diligently manage operations.

TABLE V-5
PROPOSED SAMPLING FREQUENCY FOR VARIOUS POND CHARACTERISTICS AND THE CONSIDERATIONS WHICH MAY DICTATE CHANGES TO MONITORING STRATEGIES.

	Initial Sampling Frequency	**Regular Sampling Frequency**	**Considerations**
Pond Water Quality	weekly	monthly	ore composition, milling processes, mass loadings, treatment practices, seasonal variations
Pore-waters/ Sediments	none	diagnostic monitoring only	degraded pond water quality, contaminated groundwater
Groundwater:			
Hydraulic Head	monthly	monthly/quarterly	seasonal variations (*i.e.* freshet)
Hydraulic Conductivity	once, during baseline study	none	
Chemistry	quarterly	quarterly	chemical fluctuations
Critical Depth Verification	monthly	bi-annually	depositional strategies, water balance

VI - CLOSURE MANAGEMENT AND OTHER CLOSURE OPTIONS

1.0 INTRODUCTION

The ultimate goal of a subaqueous disposal operation is to achieve a stable closure scenario. With respect to submerged tailings in a constructed impoundment,or man-made, such a state can be defined as the long-term physical and chemical stability of the tailings solids. In order to develop an appropriate closure strategy for a specific mining operation, emphasis must be placed on designing for closure throughout the mine life cycle. Specifically, the initial closure plan developed in the design stages should be continually refined throughout operations such that at mill shutdown a comprehensive final plan is already in place. The following sections address various factors that warrant attention during the pre- and post-closure phases.

2.0 PRE-CLOSURE MANAGEMENT

Prior to closure, the physical and chemical database compiled during operations should be used to establish a final closure model. The major factors on which a closure scenario are based will include the water balance, chemical reactivity, structural stability and treatment requirements. It should be recognized that the removal of various inputs to the tailings pond will dramatically alter the water balance of the pond, and therefore, the model must be adjusted accordingly.

3.0 POST-CLOSURE MANAGEMENT

Post-closure management entails the verification and implementation of the closure model established during operations. The following sections highlight relevant considerations and monitoring strategies for various aspects of closure management.

3.1. WATER BALANCE

The removal of the mill discharge from the water balance of the tailings pond will have relevance with respect to pond water depth, residence time, overflow discharge rates and treatment requirements. To satisfy management criteria, natural runoff, pond diversions, *etc.* must be engineered accordingly to meet the water requirements of the pond. Flows which were previously diverted around the pond during operations, for example, may be re-routed back into the pond in order to meet the minimum water cover criteria. An emergency spillway must be designed to accommodate the range of flows events associated with local climatic conditions.

3.2. OTHER FLOWS

Various flows may contribute to the pond water balance and water quality after mill shutdown. In addition to natural runoff, the pond may receive inputs via minesite drainages, mine/pit waters and seepages from wasterock. Some inputs may require treatment prior to discharge for a period after closure and, therefore, the pond may continue to receive treated effluents. To verify the closure design assumptions, the flows and contaminant loadings associated with these inputs should be monitored periodically.

3.3. POND WATER QUALITY

For most operations, the collection and treatment of tailings pond overflow will be required for a period after mill shutdown. To track the improvement of water quality over time and, therefore, assess the need for continued treatment, monitoring of the tailings pond supernatant should be conducted regularly after closure. Periodic sample collection will also provide an indication of *in situ* pond processes which may contribute to improved or degraded water quality. Tailings pond water samples should be analyzed for key parameters.

3.4. GROUNDWATER

Any impacts to groundwater detected during operations should be monitored to verify closure predictions and to validate any mitigative measures implemented to control them. If no operational groundwater issues presented themselves during operations, the frequency of monitoring groundwater flow characteristics and tailings characteristics should be minimal.

3.5. GEOTECHNICAL MONITORING

As for the operating period, geotechnical monitoring programs are also needed for the closure period. These closure programs are designed to ensure that all systems function in accordance with the plans developed and refined throughout the design and operating phases of a mining operation. Qualified technical specialists should be engaged to define programs in which the majority of the work may be performed by technical operations staff.

The information acquired from the monitoring programs during operations should be used to define the key parameters that identify the performance of geotechnical structures. With this information a streamlined program can be established to allow efficient monitoring on a regular but very manageable basis.

4.0 OTHER CLOSURE OPTIONS

It must be recognized that throughout the course of the life of a mine, significant technical advances will be made with regards to effluent treatment, closure technologies and monitoring tools *etc.* Consequently, the conceptual closure plan must be constantly updated particularly toward the end of the active mine life. This is best accomplished with the technical support of an active research program. It is only through active applied research that new options will be considered by other stakeholders for use in existing or new mines.

Several closure strategies that are receiving attention through on-going research are outlined below. The following section is not exhaustive but is meant to illustrate the types of approaches that may be adopted to improve a closure plan.

4.1. INTACT WATER COVER

A permanent water cover is presently one of the best approaches for enhancing the chemical stability of sulphidic mine tailings. Further enhancement of long-term stability (particularly if materials such as neutralization sludges exist in the pond) may be accomplished through the addition of substrate covers to supplement an underwater cover. Recent research has focused on the use of organic covers (*e.g.*, sewage sludge, compost) designed to consume oxygen. Inert materials (*e.g.*, non acid-generating tailings) have also received considerable interest in providing a diffusive barrier to contaminant transport. Both approaches require systems to achieve the even distribution of such solid-phases throughout the tailings facility. Another alternative is the cultivation of algae (organic layer) by controlled fertilization where primary productivity is stimulated in the tailings pond (via nutrient addition) resulting in a more even distribution of organic rich sediments over the tailings.

4.2. REMOVAL OF WATER COVER

Removal or minimization of the water cover in a constructed impoundment is one potential closure option which reduces the risk of failure of geotechnical structures. This can be done as long as potentially acid generating material is not exposed to the atmosphere. Current research is focusing on various barriers to oxygen including: non acid-generating tailings, organic sludges and soils, saturated soils and wetlands.

4.3. LAKE DISPOSAL AND SUBMARINE DEPOSITION

While this design guide has focused on constructed impoundments, it is worth acknowledging that lake or submarine disposal offers a scenario in which both chemical and physical stability may be optimized simultaneously. Lakes, natural depressions in the local geography, are implicitly stable from a geological perspective. They are not prone to geotechnical failure. Because they have a permanent water cover, they also offer the best environment for enhanced chemical stability of sulphides. Consequently, while subaqueous disposal of tailings into a natural lake imparts a short-term impact to the aquatic system, it offers significant long-term advantages to both physical and chemical stability. Similarly, submarine tailings deposition also offers a final location for tailings in a physically and chemically stable environment.

There are many complex factors that must be included in such application scenarios. Background studies and mitigative strategies are needed on a site specific basis. Effective communications with the public and regulatory authorities are also needed to obtain a full appreciation of all alternatives and the relative risks of each in both the short- and long-term.

DESIGN GUIDE FOR THE SUBAQUEOUS DISPOSAL OF REACTIVE TAILINGS IN CONSTRUCTED IMPOUNDMENTS

APPENDICES

July 6, 1999

APPENDIX A - CASE HISTORIES

A1.0 SITES STUDIED

The following is a summary of subaqueous studies conducted under the MEND Program and a brief outline of the findings. Since 1989, detailed geochemical studies have been undertaken on four natural and two constructed tailings impoundments across Canada including Anderson and Mandy Lakes in Manitoba, Benson and Buttle Lakes in British Columbia, and the Equity Silver and tailings ponds in British Columbia. Submarine tailings deposits associated with the Island Copper Mine (B.C.) and the Black Angel Mine (Greenland) were also examined. Each system studied embodies a unique range of environmental conditions which have been used as environmental end-members to ascertain the relative importance of the various environmental controls on subaqueous tailings reactivity. The subaqueous tailings deposits studied are summarized in Table A1.

A1.1 MANDY LAKE, MANITOBA

Mandy Lake is located in central Manitoba near the Saskatchewan-Manitoba border, five km south of Flin Flon. Originally a bay off the west side of the northwest arm of Schist Lake, Mandy Lake was enclosed when a causeway was built across the inlet to the Mandy Mine site. During its operations from 1943 to 1944, ore was milled on site while mine tailings were deposited directly into Mandy Lake. The tailings form a deposit that gradually slopes away from the east shore to a water depth of 1 m, before dropping off quickly into 5 m deep water. The tailings inputs consisted primarily of pyrite with 15 to 17% sulphur and appreciable quantities of zinc and copper. A study of Mandy Lake was conducted between June and September, 1990, with a focus on the geochemical behaviour of zinc, copper and lead in the tailings (MEND, 1990a,b). In this survey, two cores were collected and the pore-waters extracted. It was found that tailings were widespread in the small lake and occurred in significant concentrations in the surface sediments (top 5 mm). This appeared to be the result of continuous slumping of the tailings fan.

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

TABLE A1
SUMMARY OF SUBAQUEOUS TAILINGS DEPOSITS EXAMINED IN MEND STUDIES

System Characteristics	Mandy Lake	Benson Lake	Buttle Lake	Anderson Lake	Equity Silver Pond
Type of System	Natural Lake	Natural Lake	Natural Lake	Natural Lake	Constructed Pond
Date of MEND Study	June 1990	Aug. 1991	Sept. 1989; Oct. 1993	Spring 1989; 1991; April and Aug. 1993	Oct. 1995
Time of Tailings Deposition	Historical (1943-1944)	Historical (1962-1973)	Recent (1966-1984)	Recent (1979-1994; 1995-Present)	Recent (1980-1994)
Maximum Water Depth	5.5 m	54 m	80 m	4 m	5 m
Water Quality Rating	Good	Good	Moderate	Poor	Moderate
Biological Productivity	Mid to High Productivity	Low Productivity	Low Productivity	Mid to High Productivity	Low Productivity
Oxygen State in Water Column	Insufficient Data	Oxic	Oxic	Seasonally Anoxic	Unknown
Trace Elements Studied	Cd, Co, Cu, Pb, Zn	Cd, Cu, Pb, Zn	As, Cd, Cu, Hg, Pb, Zn	As, Cd, Cu, Hg, Pb, Zn	As, Cd, CN, Cu, Pb, Sb, Zn
Metals Influx	Zn	Zn, Pb, Cd	None	Cd, Cu, Pb, Zn	Sb
Metals Efflux	Cu, Pb	None	Cd, Cu, Pb, Zn	None	As
Solid Tailings Characteristics	Sulphidic Tailings	Sulphidic, Pyrite-rich	Sulphidic Tailings	Sulphidic, Pyrite-rich	Sulphidic Tailings
Supernatant Characteristics	Process Water	Process Water	Process Water	Process Water	Neutralization Sludges; CN-bearing Process Water
Tailings Management	None	Flocculent Added	Flocculent Added	Lime Addition	None

A1.2 BENSON AND KEOGH LAKES, BC

Coast Copper Mine discharged pyrite-rich mine tailings into Benson Lake on Vancouver Island, British Columbia, from 1962 to 1973. The lake consists of a single, tube-shaped basin, 2.2 km long, which reaches a maximum depth in the centre of approximately 54 m. The lake is fed from the eastern end by the Benson and Raging rivers. The tailings in the central basin of the lake have been covered by a veneer of natural organic-rich sediments of dominantly terrestrial origins. Benson Lake is considered to be oligotrophic, and contains soft water (MEND, 1990a, 1991, 1992).

Keogh Lake was used as a control (reference) site for this study (MEND 1992). The lake is located in an adjacent watershed and has never received any mine-related discharges. One sediment core from each lake was collected (August 1991 in Benson Lake, and November 1991 in Keogh Lake). Both the extracted pore-waters and the solid phase were analyzed for a suite of metals. The Benson core consisted of about 30 cm of “natural” (but still copper-enriched) sediments overlying copper bearing sulphide-rich tailings, depleted in zinc and lead. The Keogh core consisted of primarily organic-rich natural deposits. A thin veneer of manganese and iron-oxyhydroxide-rich material mantled the sediments in both lakes.

High dissolved iron concentrations in pore-waters below 1.5 cm depth in Benson Lake, and at 1.5 cm depth in Keogh Lake, indicated that the sediments became suboxic to anoxic at very shallow depths in both basins. High-resolution profiles of dissolved cadmium, lead and zinc in pore-waters of both lakes showed that the concentrations of these metals decreased from overlying bottom waters to the sediment. These data confirmed that at the time of sampling there was no efflux of these metals to the overlying water in either lake. Profiles in Benson Lake also showed that there was no copper efflux to the bottom waters, despite the fact that the "natural" sediments in Benson Lake contained higher concentrations of solid-phase copper than would be expected in a pristine basin.

A1.3 ANDERSON LAKE, MANITOBA

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

Geochemical assessments of Anderson lake were performed on three occasions: summer 1990, winter 1993, and summer 1993 (MEND, 1990a, 1990b, 1996b). Water column and sediment core samples (for analyses of solid-phase and pore-water) were taken from two sites to characterize under-ice and mid-summer conditions in April and August, 1993, respectively. High concentrations of dissolved iron in the shallow pore-waters indicated that the sediments become anoxic at shallow subsurface depths. There was no evidence of release of copper, cadmium or lead from the deposited tailings. In fact, decreases in metal (copper, cadmium, lead and zinc) concentrations across the sediment-water interface indicated that the sediments of Anderson Lake were acting as a sink rather than a source for these metals. Petrographic data from organic-rich natural sediments in Anderson Lake revealed significant framboidal pyrite formation, suggesting that precipitation of metal sulphides in the sediments was partially responsible for metal removal from the aqueous phase.

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

The acid-generating roadway was excavated and removed from the northern periphery of the lake in summer and fall, 1995. Since that time, there has been a significant improvement in water quality in the lake, despite the continued discharge of tailings from the floating pipeline. Dissolved zinc concentrations, for example, have decreased from ~350 µg L-1 (parts per billion) in the autumn of 1995 to <100 µg L-1 by July 1997, while the pH through the period January to July 1997 averaged 7.6.

A1.4 BUTTLE LAKE, BC

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

Preliminary studies were conducted in late 1989 and early 1990 (MEND, 1990a), which consisted of taking sediment cores and water quality samples at four stations within the lake. A second, more intensive geochemical survey was conducted in October, 1993, and involved establishing three stations to sample the water column, sediments and associated pore-waters (MEND, 1995b).

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

A1.5 EQUITY SILVER TAILINGS POND, BC

The Equity Silver mine operated from August 1980 to January 1994. In this period, tailings were discharged to the Equity Silver tailings pond via a mobile floating platform. The tailings pond has an area of approximately 1.2 km^2 and a maximum water depth of 5 m in the central portion of the pond. Tailings in the pond consist predominantly of arsenopyrite, pyrite and pyrrhotite minerals. The neutralized sludge from an ARD treatment plant was also co-disposed with the tailings. In January 1985, a cyanide leach plant was added for gold recovery.

A geochemical survey, was conducted at the Equity Silver tailings pond in October 1995 (MEND, 1996a). Replicate peepers were used to collect pore-waters from a shallow (1 m) and a deep (5 m) site within the tailings pond. Ancillary solid-phase and water column data were also collected. The distribution of most elements studied was indicative of small-scale lateral inhomogeneity. The data suggested that dissolved copper was neither released nor consumed by the tailings, while arsenic was released from pore-waters to pond waters at both sites via the dissolution apparently of As bearing iron oxyhydroxide phases and an unidentified solid phase or phases. Conversely, antimony was being consumed rapidly within the surficial deposits, presumably by adsorption to an existing solid-phase. Low (<0.08 mg/L) cyanide levels were detected in tailings pore-waters.

Significant technical advances have been made in recent years in the development and application of micro and mini-electrode technology, particularly for oxygen in the field of microbial ecology (Revsbech *et al.*, 1980; Jorgensen and Des Marais, 1990) and marine research in general (Reimers *et al.*, 1984; Reimers and Smith, 1986; Archer *et al.*, 1989; Sahami *et al.*, 1996). Used in conjunction with published molecular diffusion coefficients, dissolved oxygen concentration gradients have been used to calculate oxygen fluxes across the sediment-water interface (Jorgensen and Revsbech, 1985; Revsbech *et al.*, 1986; Gundersen and Jorgensen, 1990). Typically, high resolution dissolved oxygen data generated in this fashion have been used to assess the benthic oxygen demand and organic matter oxidation rates. The Equity Silver study represented the first application of oxygen and pH micro-electrodes in reactivity studies of submerged tailings. Direct determinations of tailings oxidation rates were calculated by measurements of dissolved oxygen from the peepers cells across the sediment-water interface.

Oxygen gradients were determined from bottom water, through the diffusive sub-layer (the stagnant layer of water column, typically a few mm thick, immediately adjacent to the sediment-water interface), across the sediment-water interface and into shallow pore-waters. Two distinct zones of consumption were observed: the first, through the diffusive sub-layer, where dissolved oxygen concentrations decreased linearly from bottom water to the sediment-water interface; and second, from the sediment-water interface deeper into the sediments, where dissolved oxygen concentrations were seen to decrease with depth, but with a concave-downward profile. The former profile was assumed to result from oxygen consumption by tailings and organic matter at the sediment-water interface, while the latter reflected oxygen consumption by tailings alone. Consequently, two approaches were used to calculate the oxidation rates. In the first case, a simple Fickian diffusion model (where the oxygen flux is directly proportional to the concentration gradient) was applied to determine oxygen consumption through the diffusive sub-layer. For the more complex second case, a first order consumption term was added to the model to account for oxygen consumption throughout the tailings deposit.

Oxidation rates were derived from each of the models and compared to published rates for other subaqueous sulphidic tailings. The oxidation rate at the sediment-water interface and at depth for a specific site in the tailings pond were 14 and 5 mg O_2 /m^2/d, respectively. The difference between the two estimates of oxidation rates reflects the recent addition of organic matter (from primary productivity) to the sediment-water interface, subsequent to cessation of tailings discharge into the pond. Because organic matter additions are only recent, the estimate of 5 mg O_2 /m^2/d derived for the sediment column is assumed to be a better representation of the actual tailings oxidation rate. This value agrees well with previously published values (utilizing other methods) of approximately 3 mg O_2 /m^2/d (David and Nicholson, 1995). These subaqueous sulphide tailings oxidation rates are lower than typical benthic oxidation rates in lakes characterized by organic-rich sediments. For example, oxygen consumption rates of natural sediments in Lake Vechen, Holland, are approximately 1 g O_2/m^2/d (Sweerts *et al.*, 1991). Subaqueous tailings oxidation rates are also significantly lower than measured subaerial sulphide tailings oxidation rates. For example, Nicholson *et al.* (1995) measured tailings oxidation rates of approximately 22 g O_2 /m^2/d, which is four orders of magnitude higher than tailings oxidation rates for sulphide-rich tailings stored under a water cover.

A1.6 THE ISLAND COPPER MINE, BC

Rupert Inlet is a wide fjord approximately 10 km long and 1.8 km wide that occupies an east-west trending, glacially-scoured trough on northern Vancouver Island. From late 1971 to early 1996, the open-pit Island Copper Mine operated on the north shore of the inlet and deposited between 33,000 and 50,000 t/d of silt- and fine-sand-sized tailings into the fjord via a submerged outfall. In total, nearly 400 million tonnes of tailings were discharged to the basin.

The low-grade porphyry-type Island Copper orebody contained approximately 0.45% Cu as chalcopyrite and approximately 0.017% Mo as molybdenite, both of which were disseminated as fracture fillings and smears on fractures and slips. The ore was milled and processed by conventional flotation, and the tailings thickened to about 45% solids by weight. The thickened slurry was pumped to a mixing chamber, mixed with twice the volume of seawater and discharged to Rupert Inlet via pipeline. The discharge point was moved at least three times during the life of the mine, but was always located between 40 and 50 m water depth (Ellis *et al.*, 1996). The tailings contained on average 300-700 ppm Cu and approximately 40 ppm Mo. Both metals occurred predominantly in sulphide crystallites occluded within silicate grains. Thus, little sulphide surface area was available for potential post-depositional chemical reaction.

Rupert Inlet is moderately deep (originally 110 to 165 m in the axial channel) and connects to the open northeastern Pacific through a narrow shallow throat (Quatsino Narrows) and Quatsino

Sound to the southwest. The original tailings disposal plan anticipated that the tailings would flow down-inlet (westward) from the outfall as a continuous density current and remain on the floor of the basin following deposition, but this expectation was thwarted by unforeseen hydrographic effects. At certain times of the year, particularly in summer, upwelling off the coast of Vancouver Island brings cold, dense water to the surface, which is transported into Quatsino Sound by tidal currents. During flood tides, the dense water plunges downward as a jet after crossing the shallow narrows. The jet occasionally has sufficient density and momentum to allow it to penetrate to the floor of Rupert Inlet whereby tailings can be resuspended up to the surface and carried out on the ebb tide into proximal Quatsino Sound. As of 1993, tailings could be detected chemically (as trace enrichments of copper) as far as 14 km southwest of the narrows.

The Island Copper subaqueous disposal operation is almost certainly the most rigorously monitored in the world. From inception in 1970 (prior to mine startup), the monitoring included extensive biological, chemical and physical surveys carried out at set, frequent intervals, ranging from daily sampling of the effluent stream to annual collection and analysis of fish, shellfish and benthos samples. Multiple stations throughout Rupert and adjacent Holberg Inlets and Quatsino Sound were occupied. Ellis *et al.*, (1996) recently reviewed the large body of data available and described a number of environmental effects, or the lack thereof, associated with the deposition:

- As anticipated, the benthos at sampling stations close to the outfall (where tailings deposition rates were highest) had an almost consistently reduced diversity. However, this effect diminished down-inlet and was not observable in Quatsino Sound.
- Fauna were rarely totally obliterated, but at the most active depositional sites, the numbers of species present fell to between 1 and 5.
- Recolonization appears to be rapid when tailings deposition ceases. For example, at some newly-stabilized sites where diversity had fallen to minimal, the inventory of benthic species rebounded within a year to number in the tens.
- Reductions of aquatic stocks (crabs, clams, mussels) were not observable in the data temporally or spatially during the operation of the mine. Ellis *et al.* (1996) conclude that there was no sign of a consistent reduction in the crab catch in Rupert Inlet due to the mine discharge.
- No trace metal contamination of organisms nor bioaccumulation was seen anywhere except in the vicinity of the concentrate loading dock where fugitive dust was implicated as the source of contamination in mussels.

The post-depositional chemical behaviour of the tailings was studied by Pedersen (1984, 1985), who showed that pore-waters in the surface sediments of both tailings and natural deposits were enriched in dissolved copper. Near-interface enrichments of dissolved copper are seen worldwide in all marine sediments, and reflect active biogeochemical cycling of the element. The dissolved copper contents seen in the near-interface pore-waters collected from the tailings were lower than the levels observed in natural muds. This suggests that the amount of copper released from the deposited tailings to the overlying bottom waters was similar to or less than what would have occurred naturally. The source of the dissolved copper enrichments was not confidently identified by Pedersen (1985), but it might have reflected the degradation of a labile copper-bearing phase such as organic matter. This option is consistent with what is now known about the inhibition of oxidation of submerged sulphides.

A different conclusion was reached regarding dissolved molybdenum (Pedersen, 1985). Pore-waters in the tailings were significantly enriched in molybdenum, by up to an order of magnitude compared to the overlying fjord waters. This implied a significant release of the metal to the inlet. The molybdenum release was attributed to the dissolution of the highly soluble alteration product MoO_3, which presumably formed during the milling of the ore. Similar enrichments of molybdenum have been seen in other tailings deposits, in particular in the Kitsault deposit in Alice Arm, B.C. (Pedersen and Losher, 1988) and may be a ubiquitous occurrence where ore contains MoS_2. The flux of dissolved molybdenum from tailings such as those in Rupert Inlet or Alice Arm does not pose an environmental concern because seawater is relatively enriched in the metal. However, this rationale does not apply to fresh waters which are typically molybdenum poor.

In summary, the large volume of submerged tailings now lying abandoned in Rupert Inlet appear to have had two major environmental impacts: the (temporary) smothering or displacement of benthos, and the (permanent) alteration of the benthic habitat. Rehabilitation of the impacted area, however, is expected to be rapid. No other long- or short-term chemical or biological effects were observed during the 26 years of monitoring data collected to date, leading to the conclusion that subaqueous tailings discharge can be practiced without contaminating the overlying water column and while maintaining fisheries in the receiving basin. The unforeseen dispersion of some tailings into Quatsino Sound, although unwelcome, has not led to negative impacts. However, this does point to the need to understand fully the hydrographic character of a receiving basin prior to finalizing the design of a submarine tailings discharge program.

A1.7 THE BLACK ANGEL MINE, WEST GREENLAND

The Black Angel massive-sulphide Pb-Zn-Ag mine operated at Maarmorilik in mid-west Greenland from 1973 to 1990. The mine site is located in a high-relief alpine landscape, 500 km north of the Arctic Circle and immediately adjacent to the 4-km long Affarlikassaa Fjord, the inner part of a fjord complex on the west coast. The ore consisted of pyrite (approximately 30%), marble (approximately 50%), sphalerite and galena, and typically graded 3.6% Pb and 10.5% Zn. The crushed ore was processed in a flotation mill at a rate of approximately 2000 t/d and separated into zinc and lead concentrates and 600,000 tonnes of tailings per year. Concentrations of a number of metals other than iron, lead, and zinc were slightly enriched in the tailings (Table A2).

TABLE A2

COMPOSITION OF THE BLACK ANGEL ORE AND TAILINGS IN VARIOUS YEARS[1]

	Ore	**Tailings**			
Year	**1988**	**1979**	**1984**	**1988**	**1989**
Pb (%)	3.6	0.3	0.15	0.2	0.2
Zn (%)	10.5	0.75	0.26	0.4	0.24
Fe (%)	13.7	20	15	16	14
Cu (mg/kg)	303	176	113	101	110
Cd (mg/kg)	484	45	17	27	21
Ag (mg/kg)	26	6	2	3.5	2
As (mg/kg)	121	77	71	93	5
Hg (mg/kg)	16	1.3	0.4	0.7	0.4

[1] Johansen *et al.*, 1991; Asmund, 1992

Affarlikassaa Fjord has a maximum depth of 65 m, and is separated by a 23-m deep sill from the 8-km long, 200-m deep Qaamarujuk Fjord immediately to the east. Prior to the start-up of mining operations, the decision was made to discharge tailings to Affarlikassaa at 30 m depth via a canted, tapered submerged outfall. The tilted outfall pipe was designed to shed air from the tailings so that secondary flotation would not be a problem, while the discharge depth was chosen to prohibit the escape of tailings to Qaamarujuk on the opposite side of the sill. Seawater was used in the flotation cells, the objective being to maintain the density of the tailings - process water slurry at a level higher than that of the receiving water in Affarlikassaa.

Before ore processing began it was anticipated that heavy metals and zinc would remain in the tailings as insoluble sulphides. Little deterioration of water quality in Affarlikassaa was expected. However, within three months of the start of production, dissolved zinc, cadmium and lead concentrations in the deep fjord waters had risen dramatically to levels far above the pre-mining natural background (Table A3). The highly basic (pH 10) process water from the mill was not the culprit (Table A3). Instead, it appeared that solubilization of the metals was occurring after the tailings had been discharged from the outfall but before they had settled to the bottom of the basin.

TABLE A3
ANALYSES OF SEAWATER AND PROCESS WATER IN VARIOUS YEARS[1]
(Dissolved metals concentrations given in µg/l)

	pH	Zn	Cd	Pb
Affarlikassaa, 1973, prior to mine startup	-	7	<0.4	1.8
Affarlikassaa, February 1974, near the outfall, 3 months after startup	-	422	2.5	200
Pristine reference station, 1974	-	<1	<0.05	0.9
Tailings process water, 1986	10	38	2.3	38
Affarlikassaa, 1986, 30-60 m depth	8	236	1.7	83

[1] Asmund, 1992

Although the mineral phases or alteration products responsible for the metal release were never firmly identified, solubilization of hydroxides or possibly carbonates probably accounted for the bulk of the metal additions to the fjord waters. The solubility characteristics of such phases are consistent with the observed chemical response of the tailings in the fjord. Metal hydroxides readily precipitate in flotation cells run at very high pH if a source of dissolved metals is present. Since $CaCl_2$ and NaCl were used in the ore-drilling brine to combat freezing, it is possible that the formation of lead-chloride complexes contributed to lead solubilization during drilling and that some of this labile lead was carried to the mill on the ore. Dissolution of lead and zinc alteration minerals (carbonates, for example) may also have occurred during milling, although no hard evidence for this is available. Lead and zinc released from such phases would immediately have precipitated as $Pb(OH)_2$ and $Zn(OH)_2$, both of which are highly insoluble at high pH. Upon discharge to seawater at pH 8, some dissolution of these hydroxide phases would have been expected. This would have enhanced the release of metals to the water column in Affarlikassaa as the tailings settled through the water column. Research by the Greenland Environmental Research Institute suggested that the main sources of the soluble metals were "...oxidized products from the sulphide minerals, probably carbonates" (G. Asmund, written communication, 1991). Such phases were never observed directly, however.

In an effort to reduce the metal release to the fjord, five conditions were imposed on the mine by regulatory authorities in the late 1970's (Asmund, 1992):

(i) The terminus of the tailings discharge pipe was to be no more than 2 m above the bottom, so as to limit the settling distance for the tailings and contact time with ambient seawater.

(ii) Air was to be removed from the tailings to minimize or eliminate post-discharge suspension.

(iii) The mill process water was to have a minimum pH of 10 to minimize lead solubility.

(iv) The density of the tailings discharge was to be kept high to minimize suspension.

(v) The tailings were to be treated chemically with Magnafloc and aluminum sulphate (both flocculating agents), and slaked lime (which assists flocculation and raises the pH).

Following adoption of these procedures in 1977, the mean dissolved metals concentrations in the fjord deep waters fell within two years from maxima of approximately 950 µ/L Zn and approximately 450 µ/L Pb to values four-fold lower. Efforts to reduce the lead concentration in the tailings by introducing improvements in ore grinding and in the use of mill reagents also appear to have contributed to the decline in the dissolved lead concentration in the fjord (Asmund *et al.*, 1994). With respect to final closure and the probability of continuing contamination of the water column, Asmund (written communication, 1991) noted that "unpublished studies indicate that the tailings will not release heavy metals if left undisturbed on the fjord bottom." As of late 1992, this prediction appeared to be accurate: zinc and lead levels in Affarlikassaa fell dramatically when mining operations ceased in July 1990. According to Asmund *et al.* (1994), this confirmed that production-related sources (the settling tailings) were the "wholly dominating" factor in the contamination of the water column.

Much of the environmental concern in the Black Angel case stemmed from the influence of the annual hydrographic cycle in Affarlikassaa. This comprises a period of stratification in the summer when fresh runoff establishes a low-salinity "lid" in the fjord, and a period of flushing of deep waters to Qaamarujuk during the winter, when sea-ice formation and consequent brine exclusion alter the density structure of the water columns in the area. Thus, lead and zinc escaped from Affarlikassaa on an annual basis. This created a relatively widespread and significant environmental problem as bioaccumulation of the metals occurred in such organisms as blue mussels and wolf-fish, both of which are consumed by natives in the region. Although wolf-fish muscle tissue in the Maarmorilik area showed no sign of increased lead burdens relative to pristine sites, the livers and bones showed significant increases during the late 1970's and early 1980's. During this period, lead levels became elevated 3 to 10 fold in the livers and 2 to 4 fold in bone. Following the decline in dissolved metal concentrations, the levels in the fish declined steadily and by 1988 were approaching those observed at the reference sites (Johansen *et al.*, 1991). By 1992, lead levels in the liver of spotted wolf-fish in the area had returned to background values (Asmund *et al.*, 1994).

The Black Angel experience reinforces the requirement that submarine tailings disposal schemes be predicated on clear knowledge of physical hydrography, ore mineralogy, appropriate milling processes and careful tailings outfall design. Failure to give due attention to any one of these variables in a disposal program can result in potentially serious negative consequences.

A2.0 SITE INTER-COMPARISONS AND INFLUENCE OF ENVIRONMENTAL PARAMETERS

A2.1 DISSOLVED OXYGEN, ORGANIC MATTER AND REDOX POTENTIAL

All of the subaqueous tailings deposits studied through the MEND program, as well as the Island Copper tailings in Rupert Inlet, share one very important characteristic: the oxidation of submerged sulphide minerals could not be linked to degradation in water quality with respect to loadings of acidity and dissolved metals. In the majority of examinations, releases of dissolved metals from sediments to overlying waters could be attributed to the natural cycling of trace elements. Effluxes generally fell within the ranges observed for pristine systems.

A second scenario common to the tailings/sediments deposits examined was that molecular oxygen content in pore waters decreased to zero at very shallow depths. Various lines of direct and indirect evidence support this observation including: (a) direct measurements of molecular oxygen concentrations in pore-waters collected with peepers (Equity Silver); (b) rapid decline of the dissolved nitrate concentration with depth in shallow pore waters (Buttle and Anderson lakes); (c) the very shallow onset of sulphate reduction indicated by analysis of sulphur species in pore waters (Benson and Buttle lakes); (d) the common occurrence of authigenic pyrite framboids in near -surface sediments (Anderson Lake); and finally (e) the appearance of dissolved manganese and particularly iron in shallow pore waters (all lakes). The latter was used in most of the MEND studies to define the oxygen-zero boundary in the sediments, the rationale being that significant concentrations of dissolved iron cannot coexist with molecular oxygen at circumneutral pH. A shallow penetration depth of molecular oxygen implies that the potential for sulphide oxidation is limited to the near-surface tailings deposits. This process effectively shields deeper layers from oxidation and is irrespective of the composition or total depth of the tailings.

Two main factors dictate the depth of the oxygen-zero boundary: the rate of oxygen consumption and the rate of resupply of molecular oxygen via diffusion to the reaction sites. The settling flux or rate of accumulation of easily degradable organic matter exerts the largest influence on the oxygen consumption rate, while the molecular oxygen concentration in bottom water coupled with the diffusive path length to the reaction site controls the resupply rate. Oxygen may also be consumed by reaction with reduced substances other than organic matter, including organics introduced to the tailings in the mill circuit. It is possible that the oxidation of sulphide surfaces could also have contributed to a portion of the oxygen depletion observed in shallow tailings in the MEND study lakes. However, the available evidence suggests that oxygen consumption largely proceeds via organic-matter associated reactions, and not via the oxidation of sulphide minerals.

The deep-water concentrations of molecular oxygen ranged widely between and within individual lakes in the MEND studies. In Anderson Lake, for example, the oxygen content near the bottom ranged from zero in late winter to near saturation in summer. Full oxygen depletion in bottom water in winter may be more common in Canadian Shield lakes than previously realized; for example, anoxia has also been observed during winter in shallow (4 m deep) Balmer Lake in northwestern Ontario (Martin, 1996). Irrespective of the profound variability in dissolved oxygen levels seen in bottom waters in the seasonal studies in the various lakes, all of the deposits showed rapid consumption of oxygen and nitrate (where measured) at shallow depths. This applies even to the constructed tailings pond at Equity Silver and suggests that the purposeful addition of "extra" organic matter to subaqueous tailings deposits may not be necessary to ensure the long-term stability of sulphide minerals.

A2.2 TIME

The shallow oxygen-zero boundary seen in the MEND studies has a welcome implication: because sulphide oxidation does not occur in the absence of oxygen (Morin, 1993; Moses and Herman, 1991), the potential tailings oxidation "window" must be confined to the immediate proximity of the sediment-water interface. Where rates of sediment or tailings deposition are high, as tends to be the case in active subaqueous tailings disposal operations, the potentially reactive surfaces of sulphide particles will be rapidly buried below the thin aerobic zone. Exposure to oxygen is fleeting under such circumstances. When discharge ceases, the exposure time will lengthen to an interval determined largely by the rate of natural sedimentation in the receiving basin. In constructed ponds where the post-closure sedimentation rate may be low, the "exposure window" might be longer and some sulphide oxidation at or near the sediment surface could be anticipated. Providing that the water cover is deep enough to prevent turbulent resuspension of sedimented particles, and assuming shallow sub-bottom anoxia, potential oxidation and metal release will be confined to a thin veneer at the interface.

In contrast to constructed ponds, the potential for a post-closure "exposure window" in a natural receiving basin is low because biological productivity and carbon-bearing fluvial inputs of organic matter would typically guarantee a shallow oxygen-zero boundary and the generation of a relatively rapid anoxic cover over the deposited tailings. In Buttle Lake, for example, such a "walk-away" condition was established within five years of cessation of the discharge.

APPENDIX B - DEFINITION OF PHYSICAL VARIABLES AND MINIMUM DEPTH METHODOLOGY

B1.0 PHYSICAL VARIABLES

B1.1 MINIMUM DEPTH CRITERIA

The following is a summary of the key physical variables associated with the development of a minimum depth criteria. The variables are presented in the order of measurable variables, variables derived from the measured ones and the output variables. These are summarized in Table IV-2.

B1.1.1 Measured Variables

Windspeed (m/s)

Windspeed records may be collected using an anemometer on a minimum of an hourly basis according to standard meteorological practices: the recording height should be 10 m above the ground surface in a clear area and should be as close to the impoundment as conditions permit. There should also be a clear line of sight between the wind station and the surface of the impoundment. Maximum daily wind readings need to be recorded as well.

Although windspeed data could be collected throughout the year, only data from those months where the impoundment is not covered by ice should be used for a bed stability assessment as a result of wave activity. During ice-cover conditions, the windspeed can be assumed to be calm as there is no transfer of energy across the air-water interface due to the presence of the ice. However, these cold-period winds are needed for the modelling of ice-contact transport of sediment due to ice build-up. The speed of the wind and its duration in units of hours can be used to estimate the parameters associated with wave development on the water surface.

Wind Direction (degrees)

Like windspeed, records of wind direction, may be collected on a minimum of an hourly basis from the instrument used to collect windspeed data. The direction of the wind can be used to determine the most common wind direction and extract the critical orientation of the pond which is used to identify the most common fetch.

Fetch Length (m)

The maximum fetch length should be determined from both the field survey of the impoundment and analysis of the most common wind direction for wave-generating winds. The fetch length and orientation of the pond will affect future construction scenarios for the impoundment and will influence the disposal pattern of material into the existing body of water contained within the pond dykes. Maximum wave heights on small impoundments (with less than 5 km fetch length) will generally be controlled by the maximum fetch length.

Tailings Gradation - median grain size (mm)

The tailings gradation is required in order to calculate the critical bed velocity required to mobilize the median grain size. Tailings samples may be available as part of the milling design or from other similar mining/milling operations. Once the mine being designed is in operation, tailings grain sizes could be analysed both in situ in the impoundment in a minimum of three different locations (shallow, moderate, deep) and as it is pumped into the impoundment. Samples could be collected at least twice per year: once after the spring thaw and again prior to ice formation to be able to identify changes in grain size as a result of the summer wind-wave activity. The samples from the impoundment could be collected with coring devices from a boat. Stratigraphy from the cores could be logged and the core divided into sub-samples for both physical and geochemical analysis. The median grain size will control the critical velocity needed to entrain the sediments.

Tailings Density (kg/m^3)

The dry weight, solids density of tailings needs to be determined from data developed as part of the milling design or from similar mining operations. Once the mine is in operation densities may be obtained from sediment samples collected from the impoundment. The density of the sediment will affect how buoyant the particles are in water and influence the critical bed velocity needed to entrain them: lower density particles will be entrained much more readily than more dense particles.

Air Temperature (^{o}C)

Air temperature needs to be collected using a thermometer for the purposes of water balance equations and evaporation potential of the site. In addition, air temperature can be used to determine thermal gradients and help estimate the flux of heat across the water-air interface which leads to the development of thermal gradients within the water body. Air temperature could be collected at the climatological station as part of a general data collection program with a minimum frequency of hourly averages and a summary daily average. Wet and dry bulb temperatures should be recorded on at least an hourly basis.

Water Temperature (^{o}C)

For large ponds it may be necessary to measure the thermal gradient within the pond on a monthly basis to monitor changes in temperature within the impoundment. The profile of temperature through the water column may be critical to determining the potential for turnover and for hyperpycnal inflow. To determine the impact of inflow, the temperature of the inflowing water should be recorded every time a discharge measurement is taken. The temperature gradient may also be used to assess the effects of seasonal freeze-up in the shallow impoundments. Temperatures could be measured every 0.2 m through the water column using a temperature probe. The time of data collection may also be noted.

Relative Humidity (%)

Values of relative humidity may be determined using a psychrometer recording on the same interval as the air temperature records. This variable is useful from the point of view of determining the water balance at the site. Relative humidity could be collected as part of a general data collection program from a climatological station.

B1.1.2 Derived Variables

Wind Duration (hours)

The storm event which produces the appropriate risk level windspeed may have a duration which is different from the duration of the same risk level rainfall event. As with rainstorm return frequencies, windspeeds with a duration of 1 hour may have different frequencies to windspeeds with durations of 2 hours or more. The dominant duration of the appropriate wind velocity may be established prior to assessment of the return frequencies of those windspeeds.

Significant Wave Height (m)

The significant wave height corresponds to the average height of the highest one third of all waves in the wave spectrum. The wave height is an expression of the energy carried by the wave form that will impact on the shore of an impoundment and influence the physical stability of the bed. It is a common design criteria used in coastal studies and coastal engineering. The significant wave height is calculated from the windspeed records for the site using the deep or shallow water equations presented in the Shore Protection Manual (USCERC, 1984). This calculation may be made during the design phase and revisited during operations and prior to closure.

Significant Wave Period (s)

The significant wave period corresponds to the average period of the highest one third of all waves in the wave spectrum. The wave period is critical to determining the stirring efficiency of the wave spectrum. The mobilization of sediments depends on the wave period. It is calculated using the deep and shallow water equations presented in the Shore Protection Manual (USCERC, 1984). As for the wave height, this calculation may be made during the design phase and revisited during operations and prior to closure.

Significant Wave Wavelength (m)

The significant wavelength is defined as the crest to crest distance for the specified significant wave period. The wavelength can be used to develop a preliminary estimate for the wave activation depth and assess the potential impact of wave motion on bed sediments.

B1.1.3 Output Variables

Near-bed Velocity (m/s)

The near-bed velocity may be computed as detailed in Part IV and compared with the critical bed velocity required to entrain the existing sediment. Near-bed velocity for the purposes of designing a tailings pond can be a parameter derived from laboratory measurements, or perhaps based on observations during the operation of a tailings pond

Minimum Required Water Depth (m)

The minimum depth of water necessary to allow a physically stable bed to form in the pond may be derived from the available climatologic and hydrologic data. The methods outlined for performing the appropriate calculations are discussed in Part IV.

Ice-Cover Thickness (m)

During operation, the thickness of the ice-cover over the impoundment in the winter may be measured on a monthly basis at a minimum of 5 locations within the pond. The ice thickness can be measured by coring through the ice and measuring the length of the core. If the ice thickness is less than the depth of water then the ice will not likely entrain sediments by freezing them. An assessment of the annual variation in ice thickness could be made to determine the thickness of ice associated with the appropriate level of risk assumed for the site.

Ice-Cover Duration (days)

Annual records of the number of days during the winter season that ice covers the impoundment could be maintained. The records may consist of those continuous periods of time that complete coverage was maintained and the number of days for freeze-up and break-up to occur.

B1.2 POND DESIGN

The following is a summary of the key physical variables associated with the development of the pond design criteria. The variables are presented in the order of measurable variables, variables derived from the measured ones, variables obtained through the site design team or design reiterations and the output variables. These are summarized in Table IV-3.

B1.2.1 Measured Variables

Precipitation (mm)

Precipitation measurements may be collected every 10 minutes using a tipping-bucket rain gauge. Precipitation measurements form part of the standard data collection suite from a climatological station. These records are used to assess the precipitation inputs into the water balance calculations for the impoundment. From the point of view of bed stability, the precipitation records are used to determine the management procedures that will be required to maintain the minimum water depth requirement.

Solar Radiation (W/m^2)

The net influx of solar radiation is needed both for the calculation of evaporation and for the thermal diffusion modelling of the pond for ice build-up. The solar radiation may be measured such that the radiative fluxes can be determined, both shortwave and longwave. Various combinations of devices can be utilized to measure these fluxes. The recording period could be over a minimum of hourly intervals and should be collected at the same location as the other climatological information.

Tailings Pond Length (m)

The maximum shore-to-shore distance across the pond may be defined as the pond length. Its magnitude and bearing should be recorded. A bathymetric profile along this line could be interpolated from the bathymetric survey information. This length could be re-evaluated each time the configuration of the pond is altered.

Tailings Pond Width (m)

The maximum shore-to-shore distance perpendicular to the length of the pond (as defined above) may be established. This distance may be defined as the pond width. A bathymetric profile along this line should be interpolated from the bathymetric survey information. This length could be re-evaluated each time the configuration of the pond is altered.

Elevation (m)

A survey may be completed which establishes the general topography of the impoundment site. This survey may be completed as part of the feasibility study and the data imported into a Digital Terrain Model (DTM) to produce a surface which can be used for design and modelling studies. This survey may be repeated in the vicinity of the impoundment should there be a significant change in the surrounding topography. The survey could be run in transects between 50 and 100 m apart with elevation points being collected at each break in slope or every 10 m whichever is less.

B1.2.2 Derived Variables

Impoundment Surface Area (m^2)

The overall impoundment geometry may be established from the site survey. In an existing pond, the geometry will have an effect on fetch direction, wave refraction, longshore transport, internal circulation and possible seiche effects. A man-made impoundment could be designed to minimize the fetch length and reduce the potential for wave refraction, longshore transport, internal recirculation and seiching.

Storm Duration (hours)

The frequency and magnitude of rainstorms depend on the duration of the storm. Rainstorms occurring over 2 hours may have different return period curves compared with rainstorms of 4 hours in length, or 6 hours in length. The dominant rainstorm duration may be determined from the available climate data prior to assessment of return period. Several rainfall storm durations may need to be tested if no regulations exist since the site may be subject to particular rainfall event characteristics which influence flood and drainage potential. The critical rainfall storm duration, usually given in hours will affect the amount of live storage the pond must be design to hold.

Evaporation (mm)

Like precipitation, evaporation data is needed to address the atmospheric outputs of the water balance calculations. It can be used from the point of view of bed stability to determine the management procedures that will be required to maintain the minimum water depth requirement. An evaporation pan may be used or the evaporation may calculated from the climatological data collected.

Seepage Volumes (m^3/s) or (L/d)

The volume of water seeping out of the impoundment will need to be quantified for the water balance calculations. These volumes should be derived directly from records of the quantity pumped or discharged out of any seepage ponds. These records will be needed to determine the management procedures necessary to maintain the minimum required water depth to reduce wave-induced suspension.

B1.2.3 Site Design Variables

Inflow/Outflow Locations

The relative locations of inflow and outflow to and from the impoundment may be established from the site survey relative to the bathymetry. Internal recirculation of water may be driven by inflow and outflow currents and may create localized areas where water velocities are sufficient to move sediment and tailings should not be deposited. The effect of the relative locations will be highly site specific. Man-made impoundments can be designed to reduce the influence inflows and outflows will have on recirculation within the water.

Processing Water Requirements (m^3/s)

The volumes of water needed by the processing plant during operation could be identified and estimated. If any of this water is directed to the tailings pond then estimates of the maximum rate at which water is discharged and of the monthly and annual volume totals should be made. During operations, records of average hourly pump rates and monthly and annual totals could be kept.

Pumped Input Volume- make up water (m^3/s)

The volume of water pumped into the impoundment will need to be quantified for the water balance calculations. During operations these volumes could be derived directly from pump records. These records will be needed to determine the management procedures necessary to maintain the minimum required water depth. The records may include average hourly discharge rates as well as monthly and annual totals.

Pumped Output Volume - decant water (m^3/s)

The volume of water pumped out of the impoundment will need to be quantified as part of the tailings pond design. During operations these volumes could be derived directly from pump records. These records will be needed to determine the management procedures necessary to maintain the minimum required water depth. The records may include average hourly discharge rates as well as monthly and annual totals.

Tailings Discharge (m^3/s)

The rate at which tailings are discharged into the pond for disposal could be estimated for the purposes of the initial design. During operations tailings flow rates and volumes are required to determine the management procedures necessary to maintain the minimum required water depth. The records may include average hourly discharge rates as well as monthly and annual totals.

B1.2.4 Output Variables

Live Storage (m^3)

The requirement for live storage is derived from the volume of water supplied by the design storm to the impoundment. The live storage volume requirement may be reduced through decanting excess water if outflow can reliably be maintained. If not, it is recommended that the live storage volume be determined for the impoundment assuming no decant option is available. The live storage volume should not infringe on the established dyke freeboard.

Freeboard (m)

The impoundment dykes may be built to a height which allows for a specified level of freeboard. In areas where the data records are short or incomplete, relatively larger freeboards could be applied than in areas where continuous, long-term data is available. The minimum required freeboard may already be established for certain areas.

Impoundment Dyke Construction Level (m)

The elevation to which dyke surrounding the impoundment should be built depends on the volume of live storage which must be accommodated on top of the water and tailings already held within the pond. The height may be derived directly from the volume of storage needed divided by the surface area of the pond.

Total Pond Storage Volume (m^3)

The total volume enclosed within the impoundment is the sum of the live storage volume, the existing water volume (determined by the minimum depth plus the evaporation depth) and the stored tailings volume. If the total projected storage volume exceeds the total existing storage volume, then the impoundment may need to be modified to accommodate the difference.

B2.0 DERIVATION OF THE MINIMUM DEPTH METHODOLOGY

The dynamics of impoundments, in general, are greatly controlled by fetch, much more so than large bodies of water such as large lakes or the ocean. However, these impoundments are still highly responsive to energy inputs from wind stress (Carter, 1991). Since impoundments are generally small compared to lakes and oceans, global scale forcing mechanisms such as geostrophic circulation tends to be negligible although seiching does occur in the largest of lakes.

A principal mechanism for erosion, transport and deposition of sediments on the bed of an impoundment is the energy associated with waves. This wave action also tends to stir oxygenated waters at the bed and mix them with bottom waters thereby increasing the potential for submerged sediments to oxidize.

The calculation of the minimum depth required to reduce wave-sediment interactions depends upon the level at which significant interaction is assumed to have occurred between the wave motion and the bed. It is commonly held that significant wave-bottom interactions do not occur until the depth/wavelength ratio (d/L) is less than 0.25. At d/L ratio of 0.5, or greater, waves are often referred to as deep water waves since the wave motion just reaches the bed. However a weak motion at the bed can be felt as far down as d/L ratio of 0.84 (Carter, 1991), which has significance with respect to the movement of fine tailings.

B2.1 WAVE PARAMETERS

The US Army Coastal Engineering Research Centre (1984) gives equations following to estimate the significant wave height, H_{sig}, significant wave period, T_{sig} and the minimum duration, t_d, required for the generation of fetch limited conditions for deep water waves.

Deep water parameters are calculated as functions of wind stress, U_w, (m/s) over the pond fetch length, F (m), as follows:

$$H_{sig} = 5.11x10^{-4} U_w F^{\frac{1}{2}}$$

$$t_d = 32.2\left(\frac{F^2}{U_w}\right)^{\frac{1}{3}}$$

$$T_{sig} = 5.93x10^{-2}(U_w F)^{\frac{1}{3}}$$

B2.2 DEPTH PARAMETER

The minimum depth calculation procedure was derived from the methodology presented by Lawrence *et al.* (1991). This computation required the input of the fetch length, F, the wind stress over the water, U_w and the threshold velocity at the bed, V_b, required to entrain the sediments to calculate the minimum required depth. The calculation they used was derived from the significant wave height, H_{sig}, the significant wave period, T_{sig} and the minimum wind duration, t_d, equations given above.

Instead of using the threshold velocity value used by Lawrence *et al.* for the Mildred Lake tailings, a critical bed velocity term, V_b, was calculated using three methods based on two equations derived from the work of Komar and Miller (1975a, b) and Dingler (1979) as well as empirical analysis of data on the incipient motion of sediments. The velocity term was input into a modified form of the equation derived by Lawrence *et al.* to produce a minimum depth calculation as a function of the fetch length, F, the wind stress over the water, U_w, the significant wave period, T_{sig} and the median grain size, D_{50}.

Lawrence *et al.* (1991) also incorporated a factor, R, in the computation to allow for the variability of wave heights during a storm. A value of R equal to 1 is associated with the average of the highest one-third of the waves, whereas a value of R equal to 1.67 estimates the minimum water depth required for the highest 1% of waves.

B2.3 THRESHOLD OF MOTION

To determine the minimum water depth required to ensure sediment stability, it is necessary to identify the critical threshold velocity that will begin to entrain sediment. For wave environments, the critical entrainment velocity is generally slower than it would be for unidirectional currents. This decrease in the entrainment velocity is due to the efficient stirring action of the wave motion.

Ideally it would be desirable to obtain threshold values based on laboratory experiments or field observations on the tailing to be impounded. However, in developing this design guide, several methods outlined in the literature regarding the calculation of threshold velocities were examined, seeking to link the critical velocity to a function of the grain size and the wave period. The three methods that were used in this study to determine the critical velocity were based on the work of Komar and Miller (1975a,b), Dingler (1979), and an empirical equation derived from data on incipient motion under oscillatory flow that have been published.

B2.3.1 Komar and Miller (1975a,b)

Komar and Miller (1973) suggested using the following formula to determine the velocity at the bed under oscillatory waves:

$$V_b = \frac{\pi H_{sig}}{T_{sig} \sinh\left(\frac{2\pi d}{T_{sig}\sqrt{gd}}\right)}$$

where V_b is the velocity at the bed in m/s, T_{sig} and H_{sig} are the significant wave period and height respectively, g is the gravitational constant (9.81 m/s^2) and d is the water depth in m.

Following from this equation, Komar and Miller (1975a,b) produced a series of equations whereby the threshold velocity for a given grain size and grain density could be determined based on the wave period and orbital diameter at the bed. In general they found that the longer the wave period the greater the orbital velocity at the bed that was required to initiate sediment motion.

Komar and Miller (1975a,b) derived two equations which may be applied to threshold of motion computations depending on whether the median size of the bed sediment is larger or smaller in diameter than 0.5 mm. For grains less than 0.5 mm in size, Komar and Miller (1973) showed that the threshold of motion is reached while the flow above the bed is still laminar while for sediments larger than 0.5 mm the threshold is reached when flow has become turbulent. In the case of mine tailings, the median grain size of the sediments is much smaller than 0.5 mm.

For those sediments smaller than 0.5 mm Komar and Miller (1975a,b) derived the following empirical relationship:

$$\frac{\rho V_b^2}{(\rho_s - \rho) g D_{50}} = 0.21\left(\frac{d_o}{D_{50}}\right)^{\frac{1}{2}}$$

where ? and ?$_s$ are the density of water and sediment respectively in kg/m^3, D_{50} is the median grain size of the bed given in cm and do is the orbital diameter of the wave motion at the bed, given in cm/s. Rearrangement this equation and the substitution of:

$$d_o = \frac{V_b T_{sig}}{\pi}$$

for the orbital diameter leads to the equation:

$$V_b = (0.21(\frac{\rho_s - \rho}{\rho})g(\frac{D_{50} T_{sig}}{\pi})^{\frac{1}{2}})^{\frac{2}{3}}$$

which can be used to determine the critical orbital bed velocity (m/s) as a function of D_{50} (mm) and T_{sig} (s).

The Komar and Miller expression for a critical threshold velocity has been revisited by others because of their definition of the "grain-size Reynolds Number" (Kamphuis 1975), however the formulation is simple to use for the purpose of initial estimates of water cover requirements.

B2.3.2 Dingler (1979)

Laboratory studies undertaken by Dingler (1979) produced empirical relationships where the wave period could be related to a function of the near-bottom orbital diameter and the grain size of the sediments. Dingler (1979) suggested using:

$$\frac{(\rho_s - \rho)gT_{sig}^2}{\rho} D_{50} = 290(\frac{d_o}{D_{50}})^{\frac{4}{3}}(\frac{\rho(\rho_s - \rho)d^3}{\gamma^2})^{-\frac{1}{9}}$$

where K is the fluid viscosity. This equation can be combined with the equation for the orbital diameter, d_o, and the rearranged to derive the following relationship:

$$V_b = \frac{\pi \frac{D_{50}}{10}}{T_{sig}}[\frac{1}{290}\frac{gT_{sig}^2}{\frac{D_{50}}{10}}(\frac{\rho_s - \rho}{\rho})(\frac{\rho(\rho_s - \rho)g\frac{D_{50}^3}{10}}{\gamma^2})^{\frac{1}{9}}]^{\frac{3}{4}}$$

where the critical orbital bed velocity (m/s) is given as function of D_{50} in mm and T_{sig} in s.

B2.3.3 Empirical Equation Derived from Threshold of Motion Data

A third method for estimating the critical velocity required to mobilize sediment was derived from the Shields Criterion and the data on incipient motion under oscillatory waves found in the literature. This method developed an empirical log-linear equation for critical velocity that was solely a function of the median grain size at the bed.

Incipient motion data for oscillatory flow used to derive the empirical linear equation was collected from Sternberg (1971, 1972), Soulsby (1983), Carter (1991), Komar and Miller (1975a, b), and Davies and Wilkinson (1979). All the published data were based on studies made over a rigid beds which may not be directly sense to submerged sediment motion.applicable to thixotropic sludge similar to the sludge modelled by Lawrence *et al.* (1991). However, the data do provide an empirical check on the other two computations and are applicable in a generic sense to submerged sediment motion.

Analysis of the published data lead to the derivation of a log-linear formula representing the critical velocity at the bed, V_b, as a function of grain size, namely:

$$V_b = 10^{(-0.72 \log D_{50} - 1.99)}$$

where V_b is expressed in m/s and D_{50} is expressed in mm. The data used to generate this equation is presented in Table B2-1. The correlation coefficient $R^2 = 0.63$ indicated a reasonably good correspondence between grain size and critical velocity but still reflects there is scatter in the data of values reported for V_b against grain size.

It must be noted that these data cover a range of grain sizes (nominally 0.001 to 1.0 mm), a range of wave periods (nominally T_{sig} between 1 and 5 s) and a range of water depths. As an additional factor, this data also represents a range of bedform conditions which have been shown (*e.g.* Brander, 1991) to cause significant variation in suspension of material but as yet cannot be accurately quantified within the context of a model of entrainment.

B2.4 PREDICTION OF THE MINIMUM DEPTH

Using the method detailed by Lawrence *et al.* (1991), the minimum depth required to maintain a stable sediment bed within a tailings pond can be estimated from the following equation:

$$d_{\min} = 0.00187\left(\frac{U_w^2 F^2}{g}\right)^{\frac{1}{3}} \ln\left[0.037\frac{R(gFU_w^4)^{\frac{1}{6}}}{V_b}\right]$$

where F is the fetch length, U_w is the wind stress over the pond, R is an empirical constant used to represent a particular portion of the waves (*e.g.* highest one third, highest one tenth) and V_b is the critical velocity at the bed. The calculation of threshold velocity was based on the three equations described above, with the equation derived by Komar and Miller (1975a,b) being selected as the most suitable of the three since the empirical equation predicted a deeper depth and the equation derived from Dingler (1979) predicted a shallower depth.

Having completed the estimate of minimum depth it is necessary to check the assumption of deep water waves that is implicit in the formulations presented. If the minimum depth is more than one-half of the wave length, which is calculated for this check as:

$$L = g\frac{(T_{sig})^2}{2\pi}$$

then the deep water assumption is applicable. If the minimum depth is less than one-half the wave length, then it may be necessary to apply the more complicated intermediate or shallow water wave theory in the analysis.

Also necessary is to bear in mind that the prediction of minimum depth depends upon the value used for V_b. Although, as suggested in this report the equation derived by Komar and Miller is useful to give an initial estimate for design, this value needs to be checked through either laboratory experiments or field experience during the operation of an impoundment prior to selecting a final water cover depth for closure of the impoundment.

B2.5 SETTLING VELOCITY AND RESIDENCE TIMES

The length of time that sediments are likely to remain in suspension needs addressing to begin to assess the impact that wave action is likely to have on the bed. The settling velocity of sediments in the 0.02-0.05 mm size range (the size range of silts and commonly of tailings) was measured to be between 0.24-1.5 mm/s for water temperatures around 50 C based on laboratory studies by Heltzel and Teeter (1987). Work by Gibbs *et al.* (1971) measured to be 6.782×10^{-4} m/s (0.68 mm/s) for spherical particles 0.034 mm in diameter under stationary water conditions.

Using the data collected by Gibbs *et al.* for the settling velocity of 0.034 mm sized grains, and reducing it by a factor of 30% to accommodate for increased fluid turbulence in moving fluids (Murray, 1970), an estimate of 4.747×10^{-4} m/s for the settling velocity in a tailing pond environment may be made. If a minimum elevation to which these silt-sized particles are suspended by wave activities is approximately 10 times the bedform height (based on resuspension observations of fine sand by Atkins (1993), then wave action over 0.01 m rippled bedforms will cause suspension of sediments to a minimum elevation of 0.10 m above the bed.

From this elevation above the bed, the median sized sediments will take a minimum of approximately 210 s (nearly 4 min) to settle back to the bottom under calm conditions. This time to settle should be treated as a minimum since the smaller sized fractions will take much longer to settle back to the bed (*e.g.* approximately 950 s, nearly 16 min, under calm conditions for grains half the size of D_{50} of the tailings). In addition, continued wave action on a time scale of the order of seconds will serve to increase the settling times further. Hence, once mobilized by waves, some of the sediments in the pond are likely to remain in suspension for prolonged periods of time.

TABLE B1
THRESHOLD OF MOTION DATA FROM LITERATURE

Velocity at Bed (m/s)	Mean Grain Size (mm)	Notes	Data Sources
0.0224	0.42	rippled bed observed	Sternberg, 1971
0.018	0.42	rippled bed extrapolated	
0.0209	0.33	observed	
0.0148	1.09	observed	
0.0327	0.33	observed	
0.0235	0.79	rippled bed observed	
0.0150	0.38	rippled bed extrapolated	
0.0197	0.38	rippled bed observed	
0.0175	0.50	ripple bed observed	
0.023	0.40	mean value	
0.15	0.10	extrapolated	Sternberg, 1972
0.36	0.01	extrapolated	
0.02	0.125	graphical analysis	Carter, 1991
0.24	0.031	graphical analysis	
0.063	0.125	graphical analysis	Soulsby, 1983
0.025	0.063	graphical analysis	
0.03	0.01	graphical analysis, T = 1s	Komar and Miller, 1975a, b
0.07	0.10	graphical analysis, T = 1s	
0.07	0.01	graphical analysis, T = 5s	
0.11	0.1	graphical analysis, T = 1s	
0.05	0.03	graphical analysis, T = 1s	
0.06	0.05	graphical analysis, T = 1s	
0.065	0.08	graphical analysis, T = 1s	
0.1	1.4	graphical analysis, T <T <30s	Davies and Wilkinson, 1979

ARD PREDICTION METHODOLOGY

Excerpt from

The Draft Acid Rock Drainage Technical Guide - Vol. 1

British Columbia Acid Mine Drainage Task Force Report
August 1989

CHAPTER 4: PREDICTION OF ACID GENERATION AND ACIDIC DRAINAGE

4.1 INTRODUCTION AND APPROACH TO PREDICTION

4.1.1 Introduction

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

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

4.1.2 Approach to Prediction

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

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

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

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

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

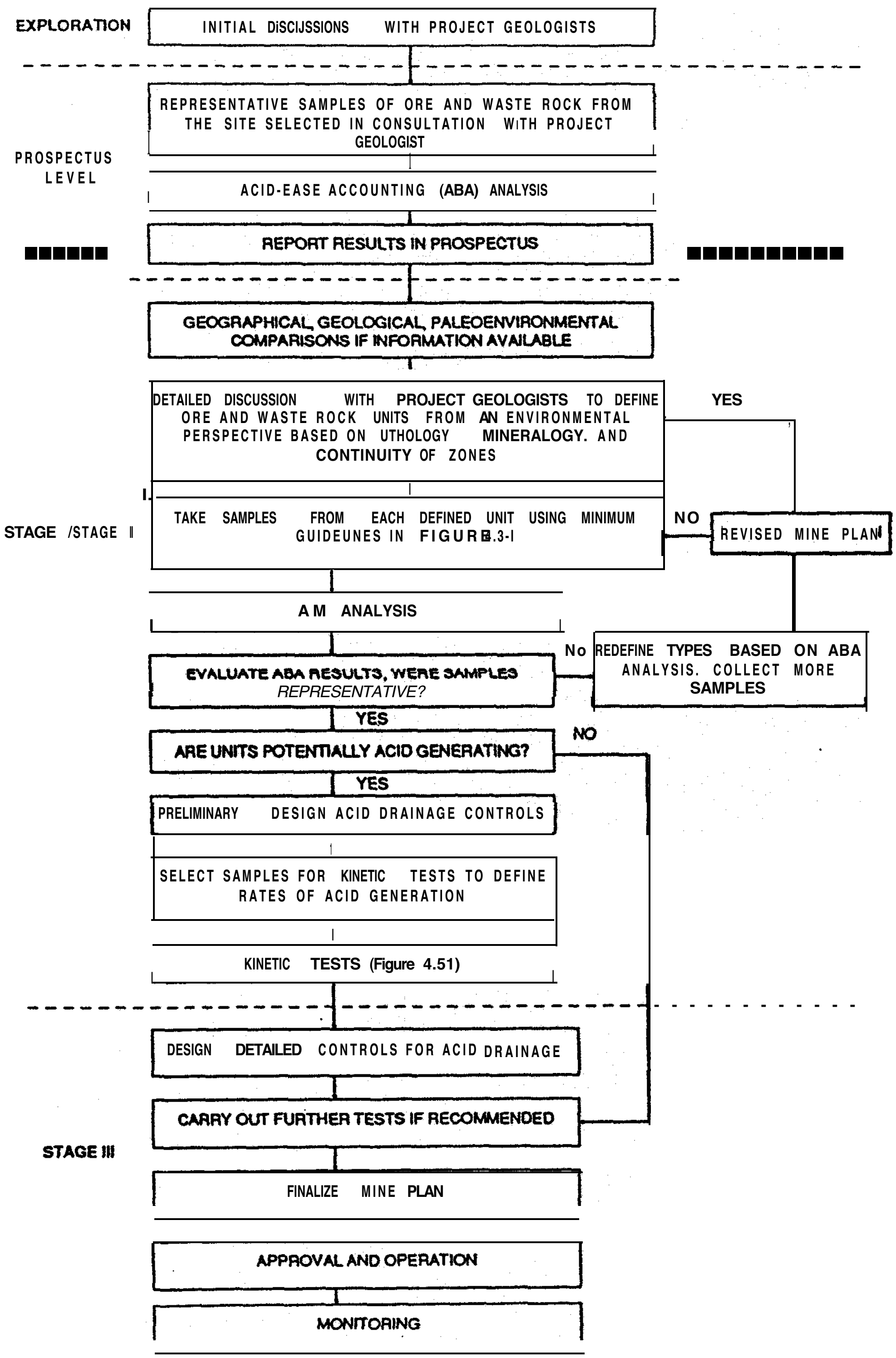


FIGURE 4.1-I
PROCEDURE FOR EVALUATING THE POTENTIAL FOR ACID GENERATION IN BRITISH COLUMBIA

4.2 GEOGRAPHICAL, PALEOENVIRONMENTAL, AND GEOLOGICAL COMPARISONS

4.2.1 Geographical Comparisons

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

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

4.2.2 Paleoenvlronmerital and geologic models

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

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

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

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

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

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

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

4.2.2.2 Examples

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

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

4.2.2.3 Summary

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

4.3 SAMPLING METHODS FOR STATIC AND KINETIC TESTS

4.3.1 Objectives of sampling

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

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

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

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

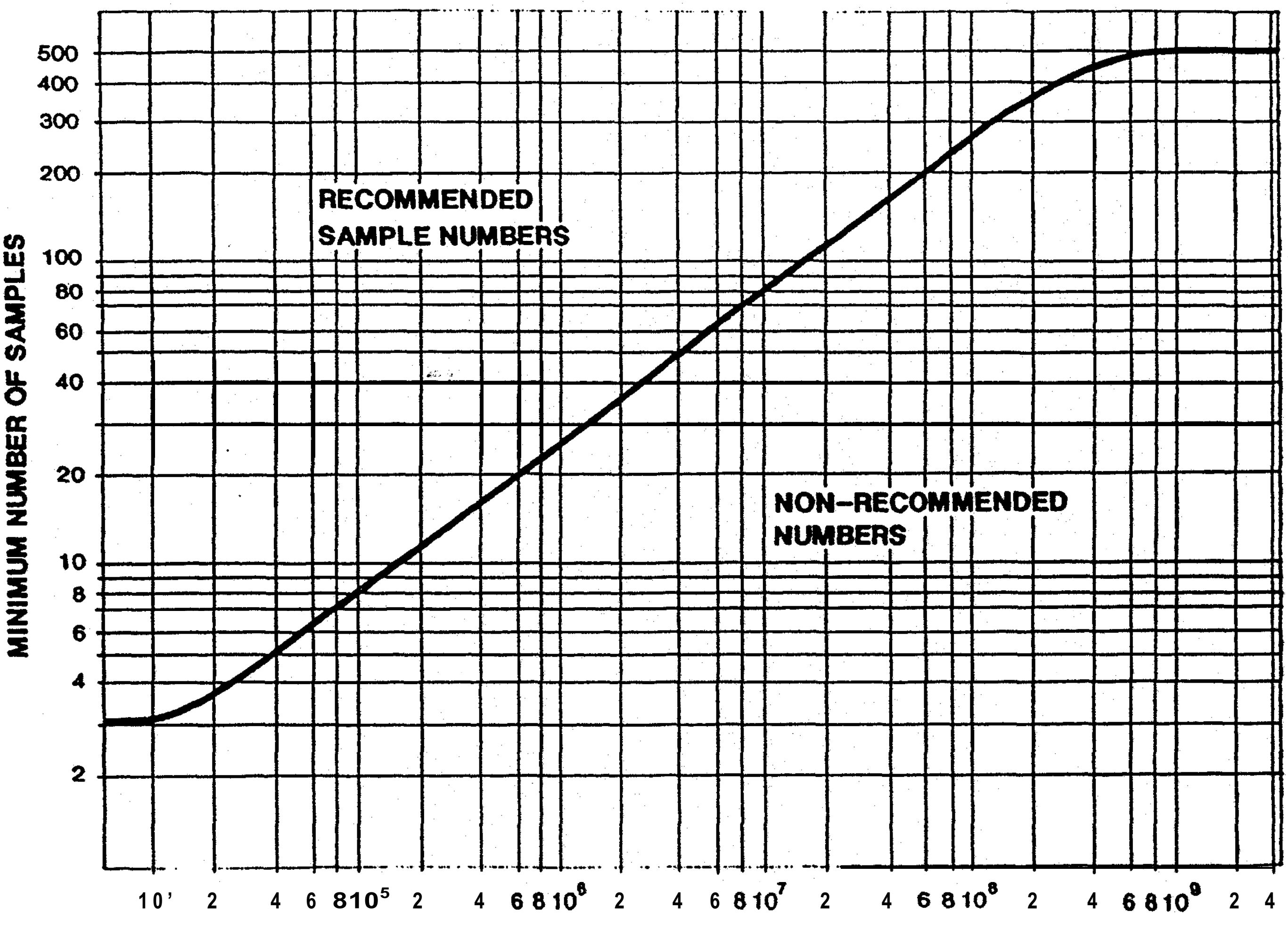


FIGURE 4.3-1
RECOMMENDED MINIMUM NUMBER OF
SAMPLES AS A FUNCTION OF MASS OF EACH GEOLOGIC UNIT
(Based on unpublished field data obtained
by Norecol Environmental Consultants)

4.3.2 Application of sampling program

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

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

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

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

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

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

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

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

4.4 STATIC PREDICTIVE TESTS

4.4.1 Introduction

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

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

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

There are several types of static tests such as acid-base accounting, APP/sulphur ratio, and the B.C. Research initial test. However, all of these tests are simply variations on a basic procedure and ail require variations of the same basic analyses (Section 4.412) **for** determining the balance between potential acidity and neutralization potential. Consequently, the basic, common procedure will be presented and the names of the variations will be de-emphasized.

4.4.2 Procedure for conducting a static test

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

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

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

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

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

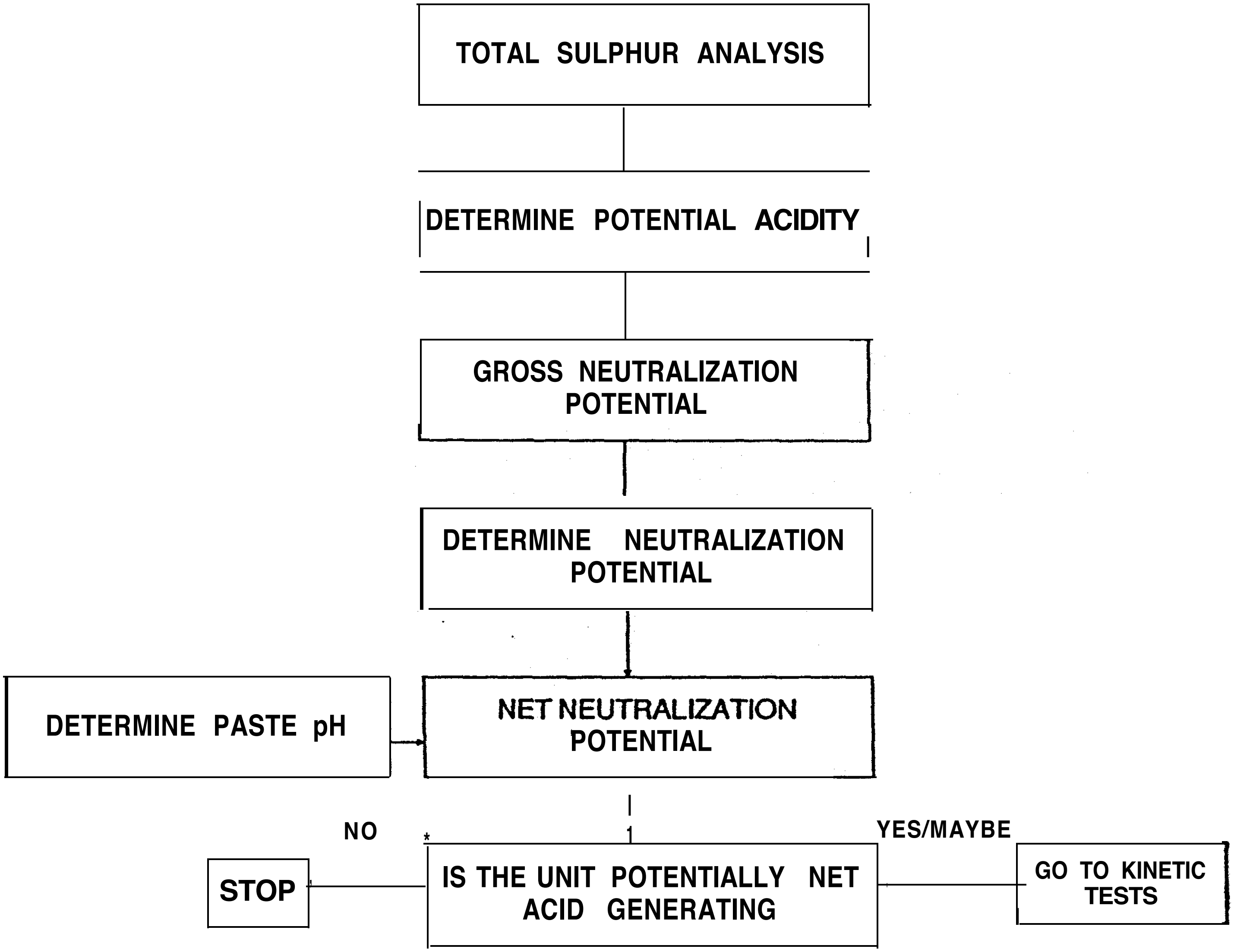
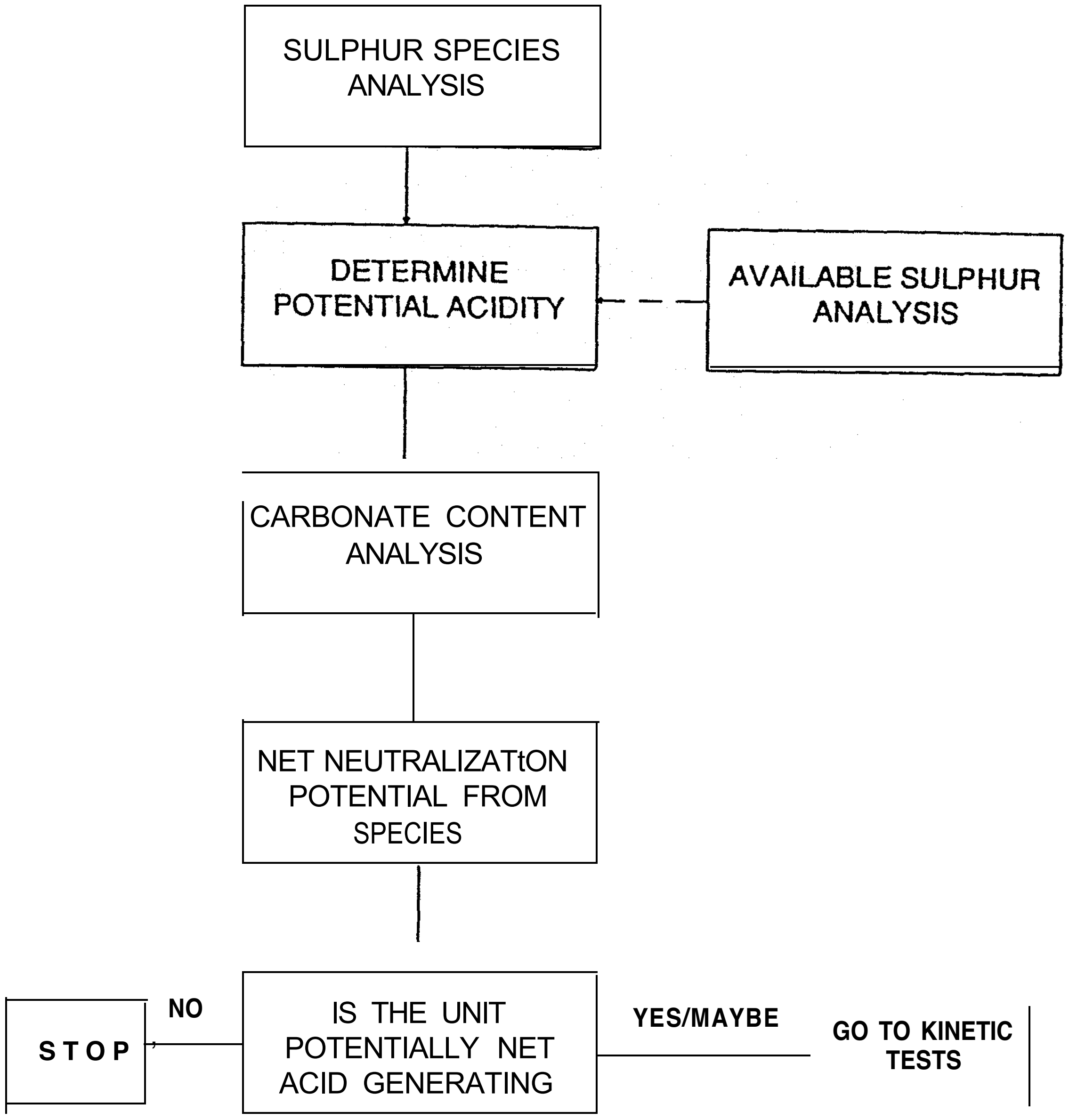


FIGURE 4.4-I
RECOMMENDED STATIC TEST PROCEDURE FOR
EACH GEOLOGIC UNIT



- -Method development required

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

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

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

4.4.2.2 intetpetation of static test results

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

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

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

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

4.5 KINETIC TESTS

4.5.1 Introduction

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

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

4.5.2 Procedures for conducting a kinetic test

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

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

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

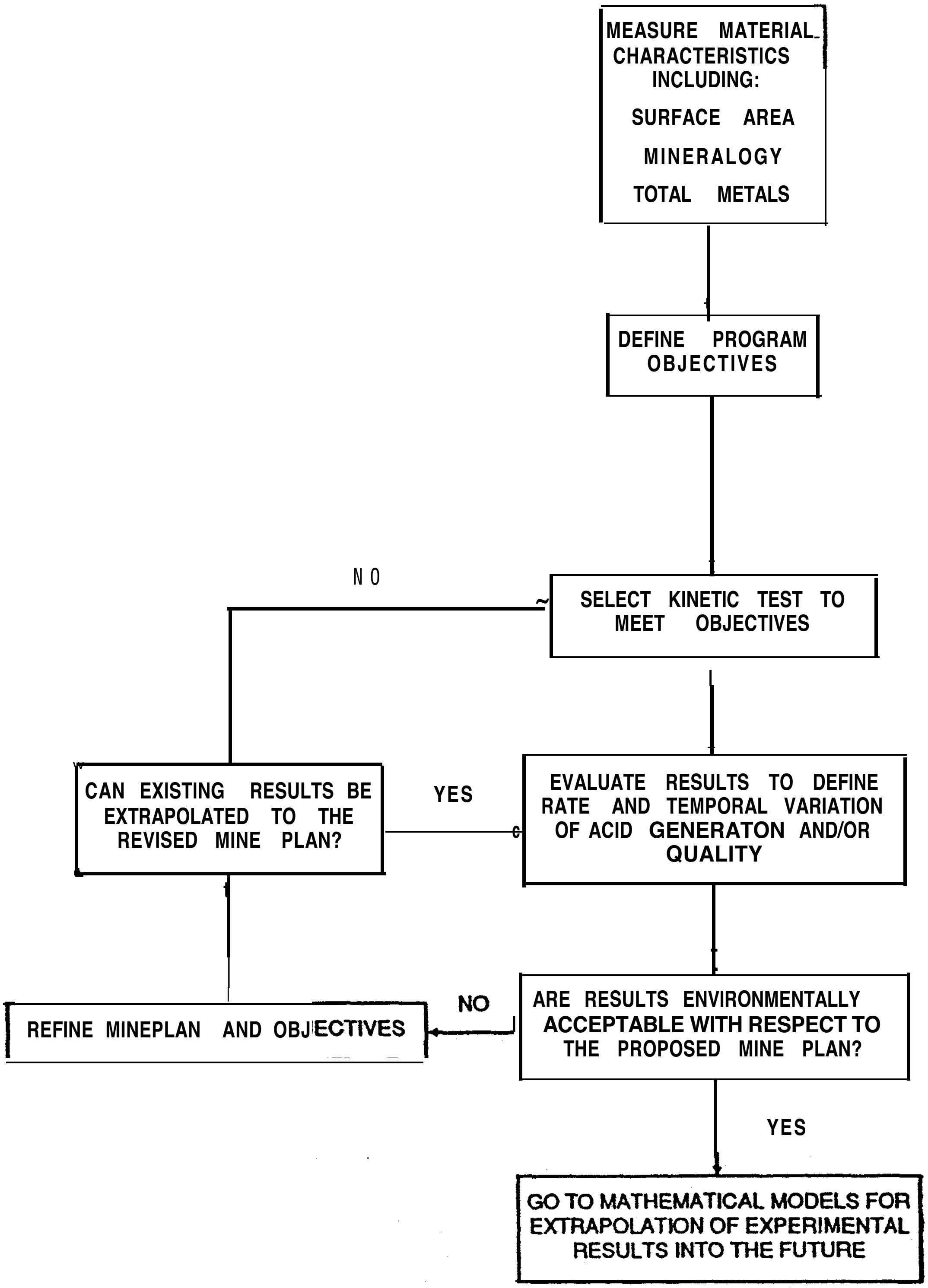


FIGURE 4.5-I
RECOMMENDED KINETIC TEST PROCEDURE

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

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

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

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

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

50th small-scale controlled tests (for example, humidity cells) and large-scale on-site weathering trials have been used in assessing acid generation reactions. The controlled tests have the advantage of simulating specific climatic and weathering conditions. On-site tests may be considered more representative than controlled tests because of the natural conditions under which the tests are conducted; however, since results vary as climatic conditions change, the interpretation and extrapolation of the test results is more complicated.

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

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

FIGURE 4.5-2
COMPARISON OF AVAILABLE KINETIC TEST METHODS

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

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

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

4.6 MATHEMATICAL MODELS

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

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

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

4.7 SUMMARY

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

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

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

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

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

APPENDIX 4.4

DESCRIPTION OF STATIC TESTS

4.4 Description of Static Test Analyses A4.4-1
- 4.4.1 Total Sulphur A4.4-1
- 4.4.2 Sulphur Species A4.4-2
- 4.4.3 Reactive Sulphur A4.4-3
- 4.4.4 Gross Neutralization Potential A4.44
- 4.4.5 Carbonate Analysis A4.4-5
- 4.4.6 Low Dectection Limit Corbonate Analysis A4.4-6
- 4.4.7 Paste pH A4.4-6

APPENDIX 4.4

4.4 DESCRIPTION OF STATIC TEST ANALYSES

4.4.1 Total sulphur

Objectives

- To determine the total sulphur content of a sample; and
- To allow the calculation of maximum potential acidity in a sample.

Description of test

Total sulphur is usually determined using a Leco sulphur analyzer. The sample is heated to approximately 1600°C with a stream of oxygen passing through the sample. Sulphur dioxide is released from the sample and collected in a solution, which is then titrated to determine the total sulphur. Total sulphur is usually reported as the percentage of sulphur relative to the entire sample (%S).

Interpretation

Total sulphur (as %S) Is converted to maximum potential acidity in units of kilograms of $CaCO_3$ equivalent/tonnes of sample (or t of, $CaCO_3$ equivalent/I000 t of sample) through multiplication by 31.25. This conversion factor is derived-as follows. Firstly, it is assumed that the pyrite is completely oxidised by oxygen and water to sulphate and solid $Fe(OH)_3$.

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

Then, it is assumed that hydrogen ions produced in the reaction are incompletely neutralized by $CaCO_3$ to a pH not greater than 6.

$$2H^+ + CaCO_3 \text{ --> } Ca^{2+} + H_2CO_3$$

Based on this reaction pair, stochiometrically the acidity produced by 1 mole of sulphur is neutralized by 1 mole of $CaCO_3$. One gram of sulphur in 100 g of material (1% S) is equivalent to 6.03125 moles of sulphur which would be neutralized by 0.03125 moles or 3.125% $CaCO_3$. This concentration is conventionally expressed as 31.25 kg $CaCO_3$/tonne of material. Thus, the conversion factor is theoretical and is based on geochemical assumptions depending on the acid-generating conditions. Realistically, the conversion factor could be significantly greater than or less than 31.25.

Advantages

- theoretical maximum amount of acidity that could be produced is calculated from total sulphur;
- the method is widely used and proven; and
- the measurement provides confirmation of cumulative quantities from other mineral specific tests. for sulphur (Appendix 4.4, Section 4.4-2, Appendix 4.5, Section 4.5.1).

Disadvantages

- acid-generating and non-acid-generating sulphur minerals are **not** distinguished; and
- rates of acid generation are not predicted.

4.4.2 Sulphur species

Objectives

- to determine the acid-leachable sulphate, sulphide and non-leachable sulphur content in a sample; and
- to allow the calculation of potential acidity in a sample **based on sulphide** content.

Description of Test

The sample is split and tests are conducted on subsamples. The first subsample is reacted, with heated hydrochloric acid (HCl), then precipitated with. barium chloride and the barium sulphate precipitate is weighted and calculated for sulphur. **This HCl-**extractable sulphur represents acid-leachable sulphate. The second subsample is reacted with nitric acid (HNO_3), then analyzed for sulphur using the same method as above. The difference between the HCl leach sulphur and the **HNO_3** leach sulphur represents leachable sulphide.

The sulphur remaining in the HNO_3 - leached subsample represents non-leachable sulphur which is stable during the time period of the leach. This subsample should be analyzed for total sulphur to confirm that all of the original sulphur is accounted for. The non-leachable sulphur may represent potentially acid-generating sulphur such as elemental sulphur and non-acid-generating sulphur such as barite (barium sulphate). An analysis for barium would indicate potential barite content.

Interpretation

The sulphide content as %S is converted to potential acidity through multiplication **with** 31.25 (see Appendix 4.4, Section 4.4.1).

Advantages

- potential acidity represents only rapidly oxidized sulphide and does not includes non-acid generating forms of sulphur.

Disadvantages

- highly reactive sulphide may be leached by the heated HCI, therefore, the HCI extractable “ sulphate” may actually be a combination of sulphate and highly reactive sulphide;
- sulphur which is initially slow to oxidize during the acid leaching, but which is capable of generating significant acidity under reasonable environmental conditions, is not detected.
- the isolated extraction of sulphate and sulphide has not been conclusively demonstrated and therefore the method requires further research and development; and
- rates of acid generation are not predicted.

4.4.3 Reactive sulphur

Objective

- to determine the amount of reactive, acid-generating sulphur contained’ in a sample.

Description of Test

The sample is first quickly prewashed with water to remove soluble sulphate. The washed material is separated into two subsamples and both subsamples are mixed with a known amount of hydrogen peroxides (40%) in water. One subsample is left for one day and the other is left for one week. After the required length of time the water is separated from the material and is analyzed for total sulphur and sulphate content.

Interpretation

The amount of reactive sulphur can be determined from the amount of sulphafe produced from the test. This reactive. sulphur is converted to potential acidity through multiplication by 31.25 (Appendix 4.4, Section 4.4.1).

Advantaqes

- potential acidity based on the highly reactive sulphur can be determined.

Disadvantaaes

- method is in development stage and has not been reliably verified;
- the potential exists for " non-feactive" sulphide' to generate acidity over extended periods of time; and
- rates of acid generation are not predicted.

4.4.4 Gross Neutralization Potential

Obiectives

- to determine the maximum neutralization potential of a sample.

Description of Test

The total amount of neutralizing minerals including carbonates and hydroxides **present** in the material is determined by treating a sample with a known excess of standardized hydrochloric acid. The sample and acid **mixture is** heated to ensure the reaction between the acid and neutralizing minerals is complete. The amount of unconsumed acid is then determined by titrating with standardized sodium hydroxide to pH 7.

Interpretation

Neutralization potential is calculated by converting the amount of base to a calcium carbonate ($CaCO_3$) equivalent, commonly in units of kg/t of sample, t/1000 t, or % carbon dioxide (CO,). This expression of neutralization potential as $CaCO_3$ is deceiving in that most natural neutralizing minerals are not capable of neutralizing pH above a value of 6. Furthermore, there is some uncertainty in the hydroxide titration in that some minerals will re-precipitate, but at varying rates, which affects the amount of hydroxide needed to reach pH 7.

Advantaqes

- the maximum amount of neutralization potential available in a **sample** may be determined; and

0 the method is widely used and has been proven to be reliable as a gross measure of total neutralization.

Disadvantaqes

0 the method does not indicate the pH to which the sample can neutralize during extended contact with acidic water; and

0 rates and extent of neutralization are not predicted.

4.4.5 Carbonate Analysis

Obiective

0 to determine the amount of strongly neutralizing, carbonate minerals in a sample.

Description of Test

Carbonate is determined by digesting the sample in an enclosed chamber with acid and heat so that carbon dioxide (CO,) gas is evolved. The CO, gas is absorbed **into** a basic solution and is measured using Leco analyzer.

Interoretation

The amount of CO, gas evolved is used to calculate the amount of carbonate present in the material. The result is converted to calcium carbonate equivalent.

Advantaaes

0 only carbonate minerals which are generally capable of neutralizing pH to 5-9 are determined; and

0 the method is widely used but is not widely used for AMD prediction and proven to be reliable.

Disadvantaaes

0 detection limit of 2 t $CaCO_3$/1000 t of sample is not low enough for predicting relatively short-term neutralization of pH, although it is sufficient for acid-base accounting purposes; and

0 cannot be used when the pyrrhotite content is greater than 10% due to the production of SO, which interferes with CO, in the Leco analyzer.

0 rates and extent of neutralization are not predicted.

4.4.6 Low detection limit carbonate analysis

Objective

- to determine the amount of strongly neutralizing, carbonate minerals in a sample.

Description of Test

The sample is placed in a glass syringe and the sample is purged with nitrogen. A strong acid is drawn into the syringe and allowed to react with the sample for several hours. The gas in the headspace with the evolved CO_2 is injected into a gas chromatograph for analysis and quantification.

Interpretation

The amount of CO_2 gas evolved is used to calculate the amount of carbonate present in the sample. The result is converted to calcium carbonate equivalent.

Advantages

- low detection limit allows prediction for short-term neutralization of pH; and

- only the carbonate minerals which are generally capable of neutralizing pH to 5-9 are determined.

Disadvantages

- rates and extent of neutralization are not predicted; and
- the method is not used routinely.

4.4.7 Paste pH

Objectives

- to determine the pH that the sample can immediately produce upon contact with water;

- to measure the capacity for immediate dissolution of calcium carbonate in a sample; and

- to indicate whether significant acid generation occurred prior to or during paste-pH analysis.

Description of Test

Paste pH is measured by a pH/reference electrode assembly and a pH meter. The electrode assembly is placed into the paste formed by mixing water and powdered rock in a specific ratio and the pH is read from the meter.

Interpretation

A paste pH greater than the pH of the mixing water indicates immediate neutralization has occurred and a pH above 7 suggests either the presence of reactive calcite or the contamination by drilling fluids. A paste pH of less than 4.0 indicates that the material contains readily available acidity from prior acid generation and the material is toxic to most plants.

Advantaues

- the pH is simple and easy to obtain; and
- the pH indicates whether the sample contains readily available acidity or alkalinity.

Disadvantaoes

- the duration of the acidic or alkaline condition of the sample cannot be predicted; and
- rates of acid or alkaline generation are not predicted.

APPENDIX 4.5

SAMPLE CHARACTERIZATION AND DESCRIPTION OF KINETIC TESTS

4.5-1 Sample Characterization A4.5-1
  4.5.1.1 Particle Size A4.5-1
  4.5.1.2 Mineralogy A4.53
  4.5.1.3 Total Metals Analysis A4.53

4.5.2 Kinetic Tests Descriptions A4.54
  4.5.2.1 BC Confirmation Tests A4.5-4
  4.5.2.2 **Soxhlet** Reactor A4.5-5
  4.5.2.3 Shake Flasks A4.5-8
  4.5.2.4 Humidity Ceils A4.5-8
  4.5.2.5 ColumrWLysimeters A4.512
  4.5.2.6 On-Site Rock Piles A4.5-15

APPENDIX 4.5

4.5.1 SAMPLE CHARACTERIZATION

4.51 .1 Particle size

Objectives

- to determine the size distribution of particles in the sample; **and**
- to allow an approximate calculation of surface area.

Description of Test

Coarse material should be dry sieved using stainless steel wire mesh; however, wit sieving may be necessary if the material contains fines. The sieves used should reflect the overall texture of the sample with screens selected from the scheme listed in Table 4.5.1-I.

Coarser screens may be added at the top end, though material finer than 0.03 mm (30 urn) cannot be sieved and should be determined using a different method such as a pipette or hydrometer method. Very coarse samples must be very large in order to be representative.

After sieving each fraction is weighed.

Calculation of Surface Area

The data may be used to determine mean diameter of the material and total surface’ area. Since the size distribution is assumed to be geometric (Table 4.5.1-1) diameters of fractions should be expressed as the log diameter (conventionally called phi). This assures that errors in measurement of the coarser fractions do not overwhelm the finer fractions.

Advantages

- permits selection of potentially problematic species for chemical analysis during a test;
- allows changes in leachate composition to be related to mineralogy.

Disadvantaaes

- the amount of metal available for leaching may be overestimated due to metal content within sample particles that are unavailable for leaching; and
- digestion of geological materials requires further research to determine the digestion of specific metals in a complex matrix.

4.5.2' KINETIC TEST DESCRIPTIONS

4.5.2.1 B.C. confirmation tests

Obiectives

- to confirm the results of static tests.

Description

The sample is placed in a flask with sufficient sulphuric acid to attain a pH of 2.5 and shaken for four hours with acid added as necessary to maintain a pH of 2.5 to 2.8. Active culture of Thiobacillus ferrooxidans is then added to the flask and the flask is weighed. The flask is then plugged with loose cotton and incubated at 35°C while continuously shaken. The pH and one dissolved metal is monitored for the first three days to ensure pH remains below 2.8 with distilled. water added to the flask to maintain constant weight. Once a pH below 2.8 is ensured, monitoring is performed every second day until microbiological activity ceases, which is defined by no further change in pH and dissolved metal concentration. At this point, one-half of the original weight of sample is added to the flask and shaken for 24 hours. If pH rises above 3.5, the test is terminated. If pH is less then 3.5, additional sample is added at one-half the original weight and shaken for 24 hours. If pH is less than 3.5 or greater than 4.0, the test is terminated. Otherwise, the flask is shaken another 48 hours and pH is recorded.

Interpretation

The objective of the test is to determine if the bacteria can oxidize enough sulphides to sulphuric acid to satisfy its acid demands. if the bacteria can remain self sustaining then there is a strong possibility that acid **mine drainage** will occur. If there is insufficient acid production, the pH will approach the natural pH of the sample above pH 3.5, then the sample is interpreted as a non-acid producer. If the pH remains in the leaching range of below pH 3.5, then there is a strong possibility that the sample is an acid producer.

Advantages

- relatively simple to use.

Disadvantages

- the test apparently confuses or ignores the concepts of solid-liquid balance, buffering and neutralization, inorganic sulphide oxidation time, the optimum pH range for T. ferrooxidans activity, and microbiological toxicity in a closed environment;
- rates of acid generation or neutralization are not predicted;
- control techniques for acid generation cannot be tested; and
- methods that have been developed recently can indicate more reliably the confirmation of acid .generation.

4.5.2.2 Soxhlet reactor

Objectives

- to confirm the results of static tests.

Description of Test

A standard Soxhlet reactor method is used with deionized water placed in the reservoir and the sample in the thimble. The water in the reservoir is heated and vaporized **and** passes into the condenser. The condensed liquid drips onto the sample in the thimble and then back into the reservoir. The leachate is analyzed after 64 to 192 hours. Variations include intermittent drying of the sample and operating the reactor at discrete intervals.

Interpretation of the Results

The water in the reservoir holds any products of acid generation. In order to confirm the results of static tests, the pH can be evaluated as follows.

pH: <3.0 = sample is strongly acid generating

3.0-5.0 = sample is acid generating and some pH neutralization may be occurring

>5.0 = sample is not significantly acid generating or neutralization of pH is overwhelming the acid generated.

Advantaaes

- results are available after a relatively short period of time;
- various control options can be tested to a limited extent; and
- the test is relatively simple to set up and operate.

Disadvantages

- results cannot be used to simulate natural conditions;
- method is used infrequently;
- the influence of bacteria on the rate of acid generation cannot be monitored: and
- boiling and refluxing of water may significantly change the geochemistry.
- method is still in the developmental stage;

4.5.2.3 Shake **flasks**

Objectives

- to determine the rate of and temporal variations in acid generation;
- to determine the temporal variation in overall water quality;
- to confirm the results of static tests; and
- to test some proposed control options.

Description of Test

The sample is placed in a flask and covered with excess water or nutrient solution. The water allows testing of effects of varying pH and water chemistry over time. Additionally, the flask can be inoculated for microbiological testing and temperature can be varied. The flask is shaken continuously and periodically sampled for water quality parameters including suiphate concentration. The sampling involves the removal of sufficient water for analysis from the flask. If the volume of the removed water is a significant portion of the total water the flask should be refilled, although this will complicate the interpretation of the test.

Interpretation of Results

The water within a flask represents a medium into which the products of acid generation accumulate through time. The monitoring of the accumulation process will indicate temporal trends in the chemical reactions. General guidelines for evaluating the test include pH, sulphate, acidity, and metals.

pH: <3.0 = sample is strongly acid generating

3.0-5.0 = sample is acid generating and some pH neutralization may be occurring;

>5.0 = sample is not significantly acid generating **or neutralization** of pH is overwhelming the acid generated.

Sulphate: Sulphate is a direct by-product of complete sulphide oxidation. As a **result,** sulphate is indicative of the rate of sulphide oxidation and acid generation if:

- all oxidized sulphur is released to the water;
- all sulphur in the water is fully oxidized to sulphate; and
- precipitation of gypsum or another sulphate mineral does **not** limit aqueous sulphate concentration.

An increase in sulphate concentrations from analysis to analysis indicates that sulphate is accumulating in the unflushed water, providing the above conditions are met. However, initial analyses may indicate only the dissolution of sulphate minerals or the release previously accumulated acid products. It is possible that sulphate may indicate a strong rate of acid generation although pH is neutral, due to the presence of carbonate minerals. Rates of acid generation and sulphide oxidation can be calculated on a weight basis (for example, mg SO_4/kg/week or mg $CaCO_3$ equivalent/kg/week), on a surface-area basis (for example mg SO_4/m^2/week) derived from surface-area- to-weight conversions (Appendix 4.5, Section 4.5.1.1), and on a cumulative basis (for example total mg SO_4/kg).

Acidity: Acidity is a cumulative measure of several aqueous species such as Fe^{2+}, $Fe(OH)^{2+}$, Al", and HSO^{4-}. As a result, acidity can be used as a gross indicator of acid generation. A correlation between sulphate and acidity frequently exists unless neutralization is significant. Rates can be calculated on a weight basis (for example, mg $CaCO_3$ equivalent/kg/week), on a surface-area basis or on a cumulative basis.

Metals: Analyses of aqueous metals will indicate the rate of metal depletion from the sample by the comparison of metal release to the original content of the metal (Appendix 4.5, Section 451.3). The release of metal from sulphide minerals during oxidation is usually not a reliable indicator of the rate of acid generation due to secondary effects such as hydrolysis and precipitation of secondary minerals.

Advantages

- rates of acid generation and neutralization in a simulate submerged, high-energy environment can be determined;
- effects of different environmental factors such as pH and temperature can be tested during the same time interval;
- temporal variations in rates of acid generation and neutralization can **be** determined:
- various control options such as limestone addition can be tested;
- influence of bacteria on the rate of acid generation may **be monitored;** and
- the test is relatively simple to set up and operate.

Disadvantages

- test is not widely used in British Columbia so reliability is undefined;
- bacteria may not acclimatize to the sample;
- high water content in flask may inhibit acid production;
- a long time may be required to complete test to meet program objectives; and
- interpretation of data is complex.

4.5.2.4 Humidity Cells

Objectives

- to determine the rate of and temporal variation in acid generation;

- to determine the temporal variation in overall water quality;
- to confirm the results of static tests; and
- to test some proposed control options.

Description of Test

The samples are placed in enclosed chambers with an entrance port for connection to the continuous flow air supply and an exit port to ensure there would be no pressurization in the cell. Pre-filtered humidified air is supplied to the cell (Figure 4.5.2-l). Distilled water is periodically added, the sample is allowed to soak for about an hour, then the water is decanted. Experience has shown that a single wash **may not be** sufficient to remove all accumulated acid products so that a second wash may be conducted and composited with the first wash. Various modifications of the technique include inoculation with an active culture of Thiobacillus ferrooxidans, wet/dry cycles, full saturation or complete submergence of a sample, and humid air **passing** directly upwards **through the** sample.

Interpretation of Results

The chemical analysis of each periodic rinse will indicate the extent of acid generation, sulphide oxidation, neutralization and metal depletion since the previous rinse. If the time between. rinses is relatively long, data must be evaluated to determine if the accumulation of acid products has slowed reaction rates. General guidelines for evaluating the test include pH, sulphate, acidity, and metals.

pH:	< 3.0 =	sample is strongly acid generating
	3.0-5.0 =	sample is acid generating and some pH neutralization may be occurring
	> 5.0 =	sample is not significantly acid generating or neutralization of pH is overwhelming the acid generated.

Sulphate: Sulphate is a direct by-product of complete sulphide oxidation. As a result, sulphate is indicative of the rate of sulphide oxidation and acid generation if:

- all oxidized sulphur is released to the water;
- all sulphur in the water is fully oxidized to sulphate; and
- precipitation of gypsum or another sulphate mineral does not limit aqueous sulphate concentration.

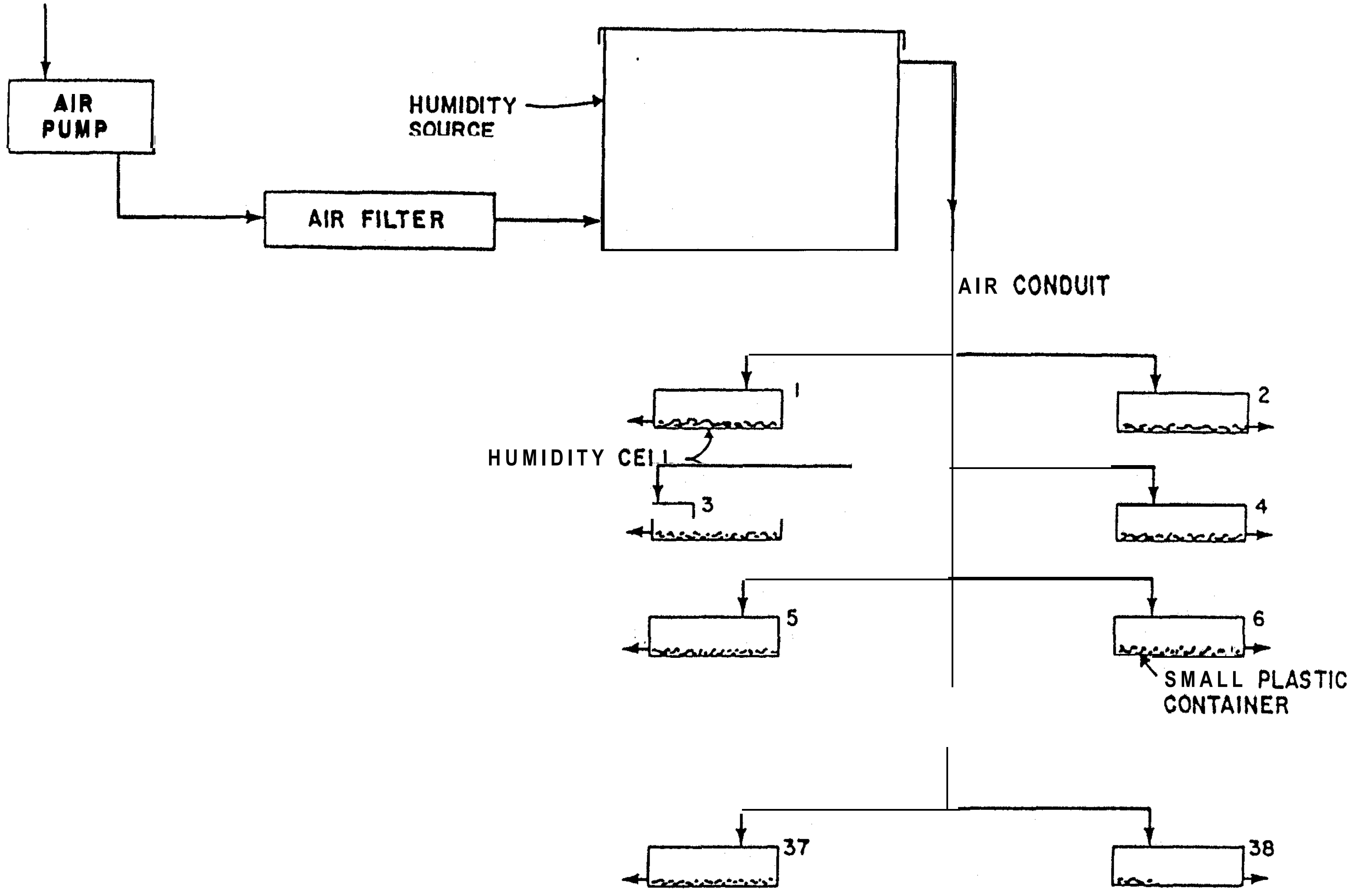


FIGURE 4.5.2-2
EXAMPLE OF A HUMIDITY CELL APPARATUS

An increase in sulphate concentrations from analysis to analysis indicates the rate of acid generation is accelerating, providing the above conditions are met. However, initial analyses may indicate only the dissolution of sulphate minerals or the release of previously accumulated acid products. It is possible that sulphate may indicate a strong rate of acid generation although pH is neutral, due to the presence of carbonate minerals. Rates of acid generation and sulphide oxidation can be calculated on a weight basis (for example, mg SO_4/kg/week or mg $CaCO_3$ equivalent/kg/week), on a surface-area basis (for example mg SO_4/m^2/week) derived from surface-area- to-weight conversions (Appendix 4.5, Section 4.5.1 .1), and on a cumulative basis (for example total mg SO_4/kg).

Acidity: Acidity is a cumulative measure of several aqueous species such as Fe^{2+}, $Fe(OH)^{2+}$, Al^{3+}, and HSO^{4-}. As a result, acidity can be used as a gross indicator of acid generation. A correlation between sulphate and acidity frequently exists unless neutralization is significant. Rates can be calculated on a weight basis (for example, mg CaCO, equivalent/kg/week), on a surface-area basis or on a cumulative basis.

Metals: Analyses of aqueous metals will indicate the rate of metal depletion from the sample by the comparison of metal releases to the original content of the metal (Appendix 4.5, Section 451.3). The release of metal from sulphide minerals during oxidation is usually not a reliable indicator of the rate of acid generation due to secondary effects such as hydrolysis and precipitation of secondary minerals.

Advantages

- rates of acid generation and neutralization in a simulated moist, unsaturated, oxygenated environment can be determined;
- as an option, rates in a simulated oxygenated, fully saturated or submerged environment can be determined;
- temporal variations in rates of acid generation and neutralization can be determined;
- various control options such as blending of waste rock can be simutated;
- influence of bacteria on the rate of acid generation may be monitored;
- the test is in use in Canada and the USA and has been compared to other predictive tests to demonstrate reliability; and
- the test is relatively simple to set up and operate.

Disadvantaaes

- bacteria may not acclimatize to the sample;
- a long time may be required to complete test to meet program objectives; and
- interpretation of results is complex.

4.5.2.5 Columns/lysimeters

Objectives

- to determine the rate of and temporal variation in acid generation on a continuous basis;
- to determine the temporal variation in overall water quality;
- to confirm the results of static tests: and
- to test treatment/mitigation options.

Description of Test

Column tests are conducted using a variety of columnar materials, such as PVC plastic, which are filled with the sample to a certain level or with a certain weight based on the specific objectives of the program. In general, a known amount of material is placed in the column and distilled water or water-based solution is flushed through the sample or ponded on top of the sample (Figure 4.5.2-2). Water samples of the flushing water or overlying pond water is periodically collected and chemically analyzed.

Some variations are:

- wet/dry cycles:
- degree of water saturation in the sample;
- fluctuating water table;
- flushing rate through the sample;
- water forced upwards through the sample; and
- inoculation with bacteria.

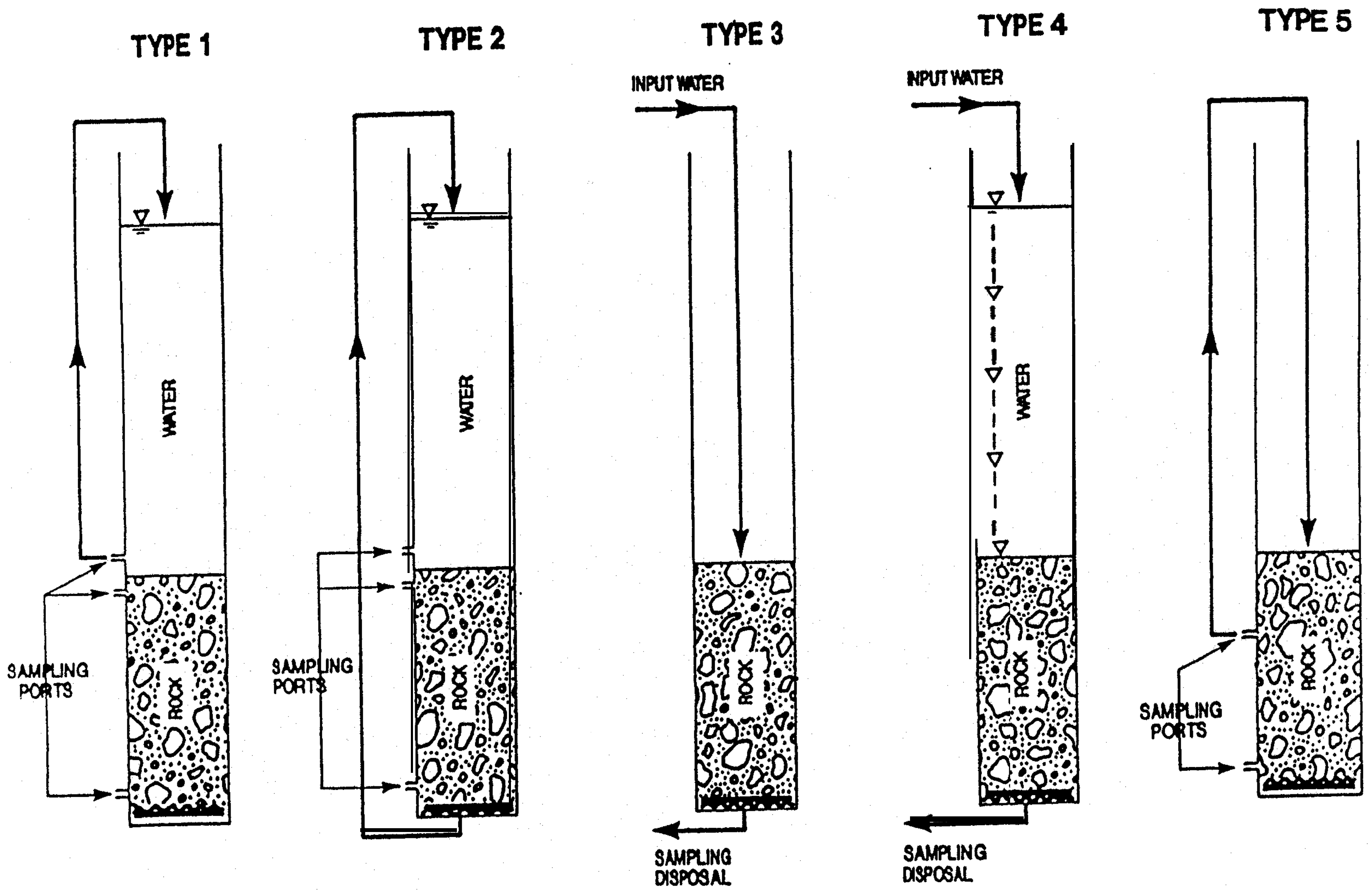


FIGURE 4.5.2-2
EXAMPLES OF COLUMN EXPERIMENTS

Interpretation of Results

The chemical analysis of water actively flushing a sample will indicate the immediate rates of acid generation, sulphide oxidation, neutralization, and metal depletion. The information can be collected on a continuous basis with an automated sample collector or periodic sampling can be extrapolated based on time between analyses and the flushing rate in the column. General guidelines for evaluating the test include pH, sulphate, acidity, and metals.

pH: <3.0 = sample is strongly acid generating

3.0-60 = sample is acid generating and some pH neutralization may be occurring

>5.0 = sample is not significantly acid generating or neutralization of pH is overwhelming the acid generated.

Sulphate: Sulphate is a direct by-product of complete sulphide oxidation. As a result, sulphate is indicative of the rate of sulphide oxidation and acid generation if:

- all oxidized sulphur is released to the water;
- all sulphur in the water is fully oxidized to sulphate; and
- precipitation of gypsum or another sulphate mineral does not limit aqueous sulphate concentrations.

An increase in sulphate concentrations from analysis to analysis indicates the rate of acid generation is accelerating of that sulphate is accumulating in ponded water, providing the above conditions are met. However, inltial analyses may indicate only the dissolution of sulphate minerals or the release of previously accumulated acid products. It is possible that sulphate may indicate a strong rate of acid generation although pH is neutral, due to the presence of carbonate minerals. Rates of acid generation and sulphide oxidation can be calculated on a weight basis (for example, mg SO_4/kg/week or mg $CaCO_3$ equivalent/kg/week), on a surface-area basis (for example mg SO_4/m^2/week) derived from surface-area- to-weight conversions (Appendix 4.5, Section 4.5.1.1), and on a cumulative basis (for example, total mg SO_4/kg).

Acidity: Acidity is a cumulative measure of several aqueous species such as Fe^{2+}, $Fe(OH)^{2+}$, Al^{3+}, and HSO_4^-. As a result, acidity can be used as a gross indicator of acid generation. A correlation between sulphate and acidity frequently exists unless neutralization is significant. Rates can be calculated on a weight basis (for example, mg $CaCO_3$ equivalent/kg/week), on a surface-area basis, or on a cumulative basis.

Metals: Analysis of aqueous metals will indicate the rate of metal depletion from the sample by the comparison of metal releases to the original content of the metal

(Appendix 4.5, Section 451.3). The release of metal from suiphide minerals during oxidation is usually not a reliable indicator of the rate of acid generation due to secondary effects such as hydrolysis and precipitation of secondary minerals.

Advantages

- rates of acid generation and neutralization under various environmental conditions can be determined;
- temporal variations in rates of acid generation and neutralization can be determined;
- various control option such as limestone addition can be simulated;
- influence of bacteria on the rate of acid generation may be monitored; and
- variations of this test are widely used and are apparently reliable.

Disadvantages

- this test is moderately complex compared to other kinetic tests;
- test may be difficult to set up and .run due to complexity of soil/water characteristics in running columns:
- bacteria may not acclimatize to the,sample;
- a long time may be required to complete test to meet program objectives; and
- interpretation of results is complex.

4.5.2.6 On-slte rock piles

Objectives

- to determine rate of and temporal variation in acid generation;
- to determine the temporal variation in overall water quality;
- to confirm the results of static tests;
- to test proposed control options;

- to determine the effect of on-site conditions on the rate of acid generation and water quality.

Description of Test

A relatively large quantity of the geologic unit to be tested is placed on an impervious basal drain at the minesite. The precipitation falling on the pile leaches through the sample and is directed to a container (Figure 4.5.2-3). At periodic intervals, the volume of seepage is measured and a water sample is collected for chemical analysis.

Interpretation of Results

The natural variability of climatic conditions at a site requires an interpretation that distinguishes the effects of the on-site climatic conditions from rates of acid generation, sulphide oxidation, neutralization, and metal depletion indicated by the chemical analysis of the water flushed through the rock pile. This is important because precipitation determines the flushing fate and the dilution of acid products, but may not affect reaction rates. General guidelines for evaluating the test include pH, sulphate, acidity, and metals.

pH: <3.0 = sample is strongly acid generating

3.0-5.0 = sample is acid generating and some pH neutralization may be occurring

>5.0 = sample is not significantly acid generating or neutralization of pH is overwhelming the acid generated.

Sulphate: Sulphate is a direct by-product of complete sulphide oxidation. As a result, sulphate is indicative of the rate of sulphide oxidation and acid generation if:

- all oxidized sulphur is released to the water;
- all sulphur in the water is fully oxidized to sulphate; and
- precipitation of gypsum. or another sulphide mineral does not limit aqueous sulphate concentration.

An increase in sulphate concentrations from analysis to analysis indicates the rate of acid generation is accelerating, providing the above conditions are met. However, initial analyses may indicate only the dissolution of sulphate minerals or the release of previously accumulated acid products. It is possible that sulphate may indicate a strong rate of acid generation although pH is neutral, due to the presence of carbonate minerals. Rates of acid generation and sulphide oxidation can be calculated on a weight basis (for example, mg SO_4/kg/week or mg $CaCO_3$ equivalent/kg/week), on a surface-area basis (for example, mg SO_4/m^2/week) derived from surface-area- to-weight conversions (Appendix 4.5, *Section 4.5.1*.1), and *on* a cumulative basis (for example, total mg SO_4/kg).

Acidity: Acidity is a cumulative measure of several aqueous species such as Fe^{2+}, $Fe(OH)^{2+}$, Al^{3+}, and HSO_4^-. As a result, acidity can be used as a gross indicator of **acid** generation. A correlation between sulphate and acidity frequently exists unless neutralization is significant. Rates can be calculated on a weight basis (for example mg $CaCO_3$ equivalent/kg/week), on a surface-area basis, or a cumulative basis.

Metals: Analysis of aqueous metals will indicate the rate of metal depletion from **the** sample by the comparison of metal releases to the original content of the **metal** (Appendix 4.5, Section 4.5.1.3). **The** release of metal from suiphide minerals during oxidation is usually not a reliable indicator of the rate of acid generation due to secondary effects such as hydrolysis and precipitation of secondary minerals.

Advantages

- rates of acid. ..generation and neutralization under on-site **climatic** conditions can be defined;
- various control options such as limestone addition can be tested;
- the test has **been** used in Canada and appears to be reliable; **and**
- **influence of bacteria on** the rate of acid generation may **be monitored.**

Disadvantages

- the test is relatively expensive and complex;
- results will vary as climatic conditions change;
- bacteria may not acclimatized to the sample;
- a long time may be required to complete test to meet program objectives; and
- interpretation of results is complex.

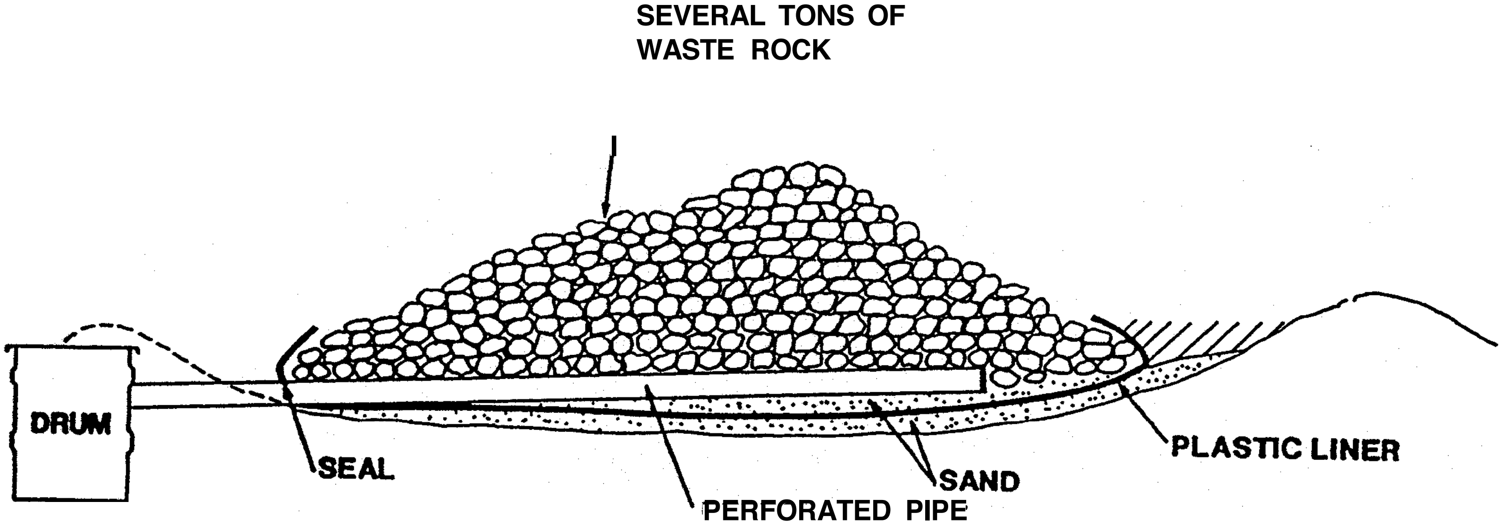


FIGURE 4.5.2-3
EXAMPLE OF A WASTE ROCK PILE

APPENDIX C - CONTRASTING BEHAVIOR OF OXYGEN IN AIR AND WATER

C1.0 THE CONTRASTING BEHAVIOR OF OXYGEN IN AIR AND WATER AND THE IMPLICATION TO SULPHIDE OXIDATION

The effectiveness of water cover in minimizing the potential for oxidation of unoxidized sulphide mine tailings is related to the behaviour of dissolved oxygen in water. In the preceding document, statements were made noting the differences in behaviour of oxygen in air and water. Following is an elaboration of the justification for those statements.

The reactivity of sulphide minerals is ultimately reliant upon the presence and availability of oxygen. In air, oxygen is both abundant and readily transported to the sulphide mineral surface and is therefore not rate limiting in its role. However, in the subaqueous environment, oxygen exists at lower concentrations and its mass transfer to reaction sites is much slower than in the atmospheric case. Consequently, sulphide oxidation is significantly retarded under a water cover.

Concentrations of solutes in solutions (gas or liquid) can be expressed in a number of ways (e.i. molarity, molality, mole fraction, parts per million). However, from a kinetic or reactivity perspective, the activity of a component is the important parameter. In the case of gasses in air (oxygen for example), activity is most often expressed by its partial pressure. Conversely, the activity of gas dissolved in water is related to its concentration. Because water and air as solvents are very different (particularly with respect to their densities), the concentrations or activities of components are not easily comparable. For example, the concentration of oxygen in air by partial pressure is 0.21 atmospheres while its concentrations by molarity and volume are 9.4×10^{-4} mol/L and 210,000 ppm, respectively. In water, by comparison, a representative concentration of oxygen at full saturation would be 3.1×10^{-4} mol/L or 10 ppm. Clearly the relative abundance of oxygen in water and air depends on how those concentrations are presented (Table C1).

TABLE C1
CONCENTRATIONS AND ENRICHMENT FACTORS OF OXYGEN IN AIR AND WATER AS EXPRESSED BY DIFFERENT MEASURES CONCENTRATION.

Unit of Measure	In Air	In Water	Concentration Factor
ppm	2.1×10^{5}	10	2.1×10^{4}
molarity (mol/L)	9.4×10^{-3}	3.1×10^{-4}	30
molality (mol/Kg)	7.82	3.1×10^{-4}	2.5×10^{4}

From the perspective of sulphide oxidation, the key issue is that regardless of how it is experessed, the concentration of dissolved oxygen is substantially lower in water than air (between 30 and 2.5 x 104 times lower depending on the expression of concentration).

What is more important in the context of sulphide oxidation, however, is the difference in the rate at which oxygen is transported or replaced in each medium. Replacement of oxygen occurs by both diffusive and advective transport; however, at the sediment-water interface, diffusion is the dominent pathway even though advection is capable of greater mass transfer. Values for oxygen diffusion coefficients in water and air are 1.5×10^{-5} and 0.178 cm^2/s, respectively, for a difference of approximately 1.2×10^{4}. In other words, once oxygen is consumed, it is replace 12,000 times more slowly in water than air. The diminutive rate at which oxygen is replaced to the site of oxidation in the subaqueous setting results in a lower degree of reactivity in water when compared to air.

C2.0 LABORATORY VERSUS FIELD STUDIES: ARE THEY COMPARABLE OR COMPATIBLE?

A number of leaching studies and incubation experiments have been carried out in different laboroatories where different types of tailings have been placed in constructed cells and submerged variously under oxygenated stirred water, stagnant water, and thin or thick water columns (e.i. Davé *et al.*, 1997). Other experimental manipulations have included the addition of a cover of organic matter on the tailings under water (e.i. Li *et al.*, 1997; Davé *et al.*, 1997). Parameters typically monitored in these experiments include pH, dissolved oxygen, and the concentrations of varous dissolved metals and sulphate.

A variety of results have been obtained from these studes, and on occasion they appear to be in conflict with observations made in the field, which has led to some confusion in the community at large. For example, Davé *et al.* (1997) noted that “small concentrations” of nickel were released from submerged pyrrhotite-bearing tailings to the overlying water in cells that he constructed and maintained for one year. The release was attributed to oxidation of Ni-bearing sulphides in the tailings, and was confined to a very thin zone right at the sediment-water interface. Oxygen depletion at very shallow depths was attributed to consumption during oxidation of sulphide particle surfaces. These results contrast with those seen in several field studies of actively depositing or abandoned tailings where very little evidence of metal release has been observed that can be unambigously attributed to oxidation of sulphide mineral phases.

To place this apparent conflict in perspective, several considerations must be kept in mind. Most importantly, and a general rule, laboratory-based observations cannot be used directly to predict the behaviour of tailings in field settings. Caution must be exercised, as putting tailings in a cell and covering them with water for an extended period does not produce a microcosm equivalent to the field. There are two main reasons for this. First, active deposition of tailings underwater at a working mine limits the average residence time of a particle at the sediment-water interace, e.i. particles are episodically if not continuously buried. Thus, their exposure time to dissolved oxygen is short, given that oxygen inventories in sediment pore waters can only be sustained by diffusion, and not by physical mixing. It is very difficult in the lab to mimic the semi-continuous or continuous burial that characterizes most submerged tailings deposits. Second, organic matter, whether in particulate form or as dissolved organic compounds, is ubiquitous in field settings, and represents a source of electrons that is readily exploitable by the bacterial community. Thus, oxygen is invariably depleted in very shallow pore waters in both lacustrine and coastal natural sediments, principally by bacterially-mediated oxidation of organic substrates, and available evidence suggests that this conclusion is equally applicable to tailings. Indeed, it is possible that similar microbial reactions could account for some of the oxygen depletion observed in tailings pore waters in laboratory studies - unfortunately, no sterile experiments have been carried out to check this supposition.

The limitations of laboratory studies have been alluded to in recent examples of such work. Li (1997), for example, note that even where metals are released in laboratory experiments, “this is not to suggest that subaqueous sulphide oxidation will cause problems at the field scale”.

Finally, this manual describes a technology for the disposal of fresh, unoxidized sulphide-bearing tailings underwater. It does not deal with the distinct issue of the disposal of oxidized sulphide bearing material. Some of the laboratory studies that have been carried out (e.i. several of the experiments reported by Dave *et al.*, 1997) have used previously-oxidized tailings as the substrate in the cells, and this distinction has been missed by some parties. Unfortunately, this has engendered some confusion. Because previously oxidized tailings have a fundamentally different chemical behaviour than fresh sulphides, the rest of the incubation studies based on altered tailings cannot be used in parallel with the information in this manual.

APPENDIX D - DATABASE COMPILATION AND APPLICATION

D1.0 INTRODUCTION

The information contained in the integrated physical and chemical database will form the foundation for management decisions regarding the need for treatment, and represents one cycle of the iterative approach required to design a tailings pond. The database can be divided generally into two components: physical and chemical. The physical aspects include those parameters associated with water cover and pond hydrography; they have direct bearing on the chemical components which consider impacts to pond water quality. Both aqueous and solid fractions impact pond water quality: the former through the direct addition of contaminants to the supernatant, and the latter indirectly through the potential dissolution of unstable solids.

Hypothetical examples of how the data are compiled and applied are presented below. All the information used (*e.g.*, metal concentrations, pond volumes) has been taken from actual mining operations. The examples illustrate calculations for both direct aqueous and indirect solid inputs to the tailings pond.

It should be noted that such calculations are not meant to model or predict the water quality of the tailings pond system. Such predictions are not practically feasible since there is no way to account for complex geochemical or dilutionary mechanisms that may occur. Rather, this approach is meant to provide a "scoping tool" to determine where high priority problems may exist. If a potential contaminant source is identified at an early stage in a project's development, proactive measures may be undertaken to avoid costly treatment scenarios at a later date.

D2.0 DIRECT AQUEOUS INPUTS

Direct aqueous inputs come from many sources. These include natural surface inflows, tailings slurry water, precipitation, waste rock seepage. The relative importance of each input will depend on its contaminant loading and rate of input to the pond. Different mining operations will have different combinations of inputs to their tailings pond. For the purposes of design, these inputs are estimated from the mine plan, relevant testwork and baseline environmental data. The contribution from each of these inputs can be represented as mass loadings by multiplying the measured or predicted flows by the contaminant concentrations to give total loadings per unit time. The mass loadings can then be compared to one another to identify which, if any, are potentially the most problematic.

D2.1 TAILINGS SLURRY WATER

A proposed underground copper-zinc mine has been designed to operate at a tailings discharge rate of 1,400 t/d. Operating at 35% solids with an average density of 2.8 g/cm^3, the mill plans to discharge approximately 5.6 x 10^5 L/d of slurry water to the tailings pond. Pilot plant test results on a number of ore samples, used in conjunction with short-term tailings stability tests, yielded an average composition of 15 µg/LZn and 7 µg/L Cu. In this example, the mill operates at pH 10, and as a result, dissolved metal levels are relatively low. The mass loadings of these elements translate to:

For Zn: 0.015 mg/L x 5.6 x 10^5 L/d = 3.1 kg/y
For Cu: 0.007 mg/L x 5.6 x 10^5 L/d = 1.4 kg/y

However, because process water is to be taken directly from the tailings pond, the metal loadings occur with no net addition of water.

D2.2 WASTE ROCK DRAINAGE

The hypothetical minesite in question occupies 55 hectares. Baseline precipitation data and the local geology suggest that the pond will receive approximately 7.0 x 10^7 L of annual runoff from the minesite. Twenty-three hectares of the minesite (42% of the minesite area) will serve as a temporary waste rock storage site, the drainages from which will flow into the tailings pond untreated. Host rock stability testwork indicates that drainages from the waste rock area can be expected to host an average of 1.0 mg/L Zn and 0.5 mg/L Cu. Metal concentrations in the other natural runoff inputs were predicted to be negligible. Assuming an annual runoff of 2.9 x 10^7 L (e.i., 42% of 7.0 x 10^7 L)from the waste rock area the mass loadings for Zn and Cu work out to:

For Zn: 1.0. mg/L x 2.9 x 10^7 L/d = 29.0 kg/y
For Cu: 0.5 mg/L x 2.9 x 10^7 L/d = 14.5 kg/y

D2.3 MINE WATER

Baseline meteorological and geological data predict approximately 1.2 x 10^6 L/d of acidic metal-rich mine waters will have to be pumped to the tailings pond. The mine waters will be discharged with the tailings stream (pH 10) and will therefore be neutralized before discharge to the pond. However, during periods of mill shutdown (2 days per week) mine waters will continue to be pumped to the tailings pond. Experimental data suggest that these unbuffered mine discharges will host 3.0 mg/L Zn and 1.0 mg/L Cu. Assuming loadings of dissolved metals will be only be significant 2 days/week (*i.e.*, 104 d/y), averaged loadings over a year approximate to:

For Zn: 3.0 mg/lL x 1.2 x 10^6 L/d x104d/y = 374 kg/y
For Cu: 0.5 mg/L x 1.2 x 10^6 L/d x 104d/y = 62 kg/y

D2.4 MINESITE DRAINAGE

The natural runoff from the remaining 32 acres (58% of the area) of the minesite is predicted to be only minimally contaminated from residues of products such petro-chemicals used in mining operations. Concentrations of zinc and copper in this natural runoff are predicted to be approximately 0.02 and 0.05 mg/L, respectively. Assuming the impoundment receives approximately 4.1 x 10^7 L of annual runoff (or 1.1 x 10^5 L/day) from the minesite, the mass loadings translate to:

For Zn: 0.02 mg/l x 1.1 x 10^5 L/d = 0.8 kg/y
For Cu: 0.05 mg/l x 1.1 x 10^5 L/d = 2.0 kg/y

The predictive calculations above indicate that unbuffered mine waters present a major potential source of dissolved metals and acidity to the tailings pond.

D2.5 NATURAL SURFACE FLOW

The tailings pond intercepts a natural stream with an annual flow of approximately 6.7 x 10^6 m^3/yr and Zn and Cu concentrations of 2 and 1 Fg/L, respectively. Despite the comparitively large inflow of dilutionary water, the natural inflow contributes mass loadings of:

For Zn: 0.002 mg/L x 6.7 x 10^9 L/y = 13.4 kg/y
For Cu: 0.001 mg/L x 6.7 x 10^9 L/y = 6.7 kg/y

D3.0 INDIRECT (SOLID PHASE) INPUTS

Tailings solids on the pond floor may contribute to pond water quality through the dissolution of solids and subsequent diffusion of metals across the sediment-water interface into bottom waters. The data from the subaqueous stability testwork can be applied to the pond design by first compiling a list of solid-phase materials proposed to be disposed of in the pond. The database would also provide information on the anticipated efflux associated with each solid (from the geochemical testwork), the rate and volume of input as well as the initial strategy of deposition (*i.e.*, will areas of a particular deposit be exposed to the water column for a long period of time or will it be buried rapidly). The anticipated fluxes (if any) are then adjusted for exposed surface area in the pond and converted to a mass loading.

Using tailings generated in a pilot plant, long-term subaqueous reactivity testwork was conducted in order to predict the post-depositional reactivity of the tailings in the pond. From the experiment, diffusive effluxes of zinc and copper were measured to be 8 and 6 $\mu g/m^2/d$, respectively. These fluxes represent the magnitude of transport of a dissolved metal from the tailings to the overlying waters. Assuming a tailings surface area of 1.2×10^6 m^2, the fluxes of zinc and copper correspond to mass loadings of:

For Zn: $8\ \mu g/m^2/d \quad x \quad 1.2 \times 10^6\ m^2 \quad x \quad 365\ d \quad = \quad 3.5\ kg/y$

For Cu: $6\ \mu g/m^2/d \quad x \quad 1.2 \times 10^6\ m^2 \quad x \quad 365\ d \quad = \quad 2.6\ kg/y$

Assuming the same pond residence time of 182 d and a pond volume of 3.6×10^9 L, such loadings correspond to increases in the pond concentrations of Zn and Cu by 0.5 µg/L and 0.4 µg/L, respectively. In this example, therefore, inputs to the dissolved metal inventory in the pond from the tailings solid phases are predicted to be insignificant.

D4.0 DATABASE APPLICATION

In terms of the relative impacts of all three inputs to pond water quality, a crude calculation can be made which takes into account the residence time and volume of the pond. It should be stressed that this over simplified one-box model is designed only to illustrate the relative impacts each input may have on pond water quality, and does not take into account other variables which will dictate the overall pond condition (*e.g.*, chemical reactivity with the pond, seasonal variability, dilution effects). Rather, the example illustrates the relative impacts of the various sources of contaminants to the tailings pond and provides a mechanism whereby proactive refinements to the mine plan or treatment strategies may be made in advance of project finalization.

For the sake of simplification, the calculation below assumes a pond steady state; *i.e.*, the rate of input equals the rate of output. The increase in concentration of zinc or copper in the pond can therefore be estimated to be:

$$Concentration_{final} = Mass\ Loading\ x\ \frac{Pond\ Residence\, Time}{Pond\ Volume}$$

The predicted pond concentration resulting from the respective inputs and the cumulative final pond concentrations are presented in Table D-1. The estimates are based on a pond dimension of 1,200 x 1,000 x 3 m (3.6×10^9L) and a residence time of 6 months (182 d) as determined from the sum of the net inflows divided by the volume of the impoundment.

Once all inputs to the pond have been identified and assessed, water quality can be predicted to determine whether the present pond configuration and mine plan fit within an appropriate environmental framework (*i.e.*, assimilative capacity of receiving environment). Decisions can then be made as to whether water treatment is necessary, or if modifications to the mill circuit, tailings management plan or pretreatment strategies are warranted. In the example above (summarized in Table D-1), the most significant input can be seen to arise through the introduction of unbuffered mine water to the tailings pond. Having performed the mass loading calculations, a management decision can now be made as to whether pre-treatment of mine waters is warranted or whether the elevated copper and zinc concentrations are tolerable in the pond, given predicted dilutive and assimilative capacities of the receiving environment.

TABLE D1
PREDICTED LOADINGS AND RESULTING POND CONCENTRATIONS OF ZINC AND COPPER IN A HYPOTHETICAL TAILINGS POND.

	Zinc		**Copper**	
	Mass Loading (kg/y)	**Predicted Pond Concentration (μg/l)**	**Mass Loading (kg/y)**	**Predicted Pond Concentration (μg/l)**
Tailings Supernatant	3.1	0.4	1.4	0.2
Waste Rock Drainage	29.0	4.0	14.5	2.0
Mine Water	374.0	51.9	62.0	8.6
Mine Site Drainage	0.8	0.1	2.0	0.3
Natural Runoff	13.4	1.9	6.7	0.9
Tailings Solids	3.5	0.5	2.6	0.4
Cumulative Loading/ Concentration	423.8	58.8	89.2	12.4

APPENDIX E - SCIENTIFIC/TECHNICAL REFERENCES

E1.0 MEND RESEARCH REPORTS

ASL Analytical Services Laboratories Ltd., 1993. Laboratory Analysis and Quality Assurance/Quality Control for the MEND Project - Anderson Lake Winter Survey, University of British Columbia Department of Oceanography, Vancouver, B.C.

Fraser, W.W stat., and J.D. Robertson, 1994. "Subaqueous Disposal of Reactive Mine Waste: An Overview and Update of Case Studies - MEND/Canada", Proceedings of the Third International Conference on the Abatement of Acidic Drainage, Pittsburgh, PA.

Hay & Co. Consultants Inc., 1996. Shallow Water Covers - Equity Silver Base Information on Physical Variables, Mend Report 2.11.5ab.

INRS-EAU, 1993. Literature Review Report: Possible Means of Evaluating the Biological Effects of Subaqueous Disposal of Mine Tailings, MEND Report 2.11.2a.

Maynard, A.W., ASL Ltd., 1994. "The Laboratory Component of Environmental Monitoring: A Case Study Involving Subaqueous Tailings Research", Proceedings of the Third International Conference on the Abatement of Acidic Drainage, Pittsburgh, PA.

Pedersen, T.F., *et al*., 1991. "Diagenetic Reactivity of Mine Tailings in Mesotrophic and Oligotrophic Lakes in Manitoba and British Columbia", Proceedings of the Second International Conference on the Abatement of Acidic Drainage, Montreal, P.Q.

Pedersen, T.F., *et al*., 1994. "Geochemistry of Reactive Tailings in Anderson Lake, Manitoba, and Buttle Lake, British Columbia", Third International Conference on the Abatement of Acidic Drainage, Pittsburgh, PA.

Rawson Academy of Aquatic Science, 1992. A Critical Review of MEND Studies Conducted to 1991 on Subaqueous Disposal of Tailings, MEND Report 2.11.1d.

MEND, 1988. Underwater Disposal of Waste Rock and Tailings, MEND Report (No Number). Rescan Environmental Services Ltd.

MEND, 1989. Subaqueous Disposal of Reactive Mine Wastes: An Overview, MEND Report 2.11.1a. Rescan Environmental Services Ltd.

MEND, 1990a (March). (4 Parts) A Preliminary Assessment of Subaqueous Tailings Disposal in (1) Buttle Lake, B.C., (2) Anderson Lake, Man., (3) Benson Lake, B.C., (4) Mandy Lake, Man., MEND Report 2.11.la. Rescan Environmental Services Ltd.

MEND, 1990b (September). (2 Parts) Geochemical Assessment of Subaqueous Tailings Disposal in (1) Mandy Lake, Man., (2) Anderson Lake, Snow Lake Area, Man., MEND Report 2.11.1b. Rescan Environmental Services Ltd.

MEND 1991. A Preliminary Biological and Geological Assessment of Subaqueous Tailings Disposal in Benson Lake, B.C., MEND Report 2.11.1c. Rescan Environmental Services Ltd.

MEND 1992. Chemical Diagenesis of Submerged Mine Tailings in Benson Lake and Natural Sediments in Keogh Lake, Vancouver Island, B.C., MEND Report 2.11.1c. Rescan Environmental Services Ltd.

MEND 1995a. Subaqueous Tailings Geochemistry: An Overview of the Reactivity of Subaqueous Mine Tailings Deposit, prepared for Placer Dome Inc. Rescan Environmental Services Ltd.

MEND 1995b. Geochemical Assessment of Subaqueous Tailings Disposal in Buttle Lake, B.C., 1993 Study Program, MEND Report 2.11.4a. Rescan Environmental Services Ltd.

MEND 1996a. Geochemical Assessment of Equity Silver Tailings Pond, MEND Report 2.11.5c. Rescan Environmental Services Ltd.

MEND 1996b. Geochemical Assessment of Subaqueous Tailings in Anderson Lake, B.C., 1993-1995 Study Program, MEND Report 2.11.3abc. Rescan Environmental Services Ltd.

Sly, P., 1996. Review of MEND Studies fro 1992 to 1995, MEND Report 2.11.1e.

E2.0 CITATIONS

Allen, J.R.L.,(1982). “Sedimentary Structures: Their Character and Physical Basis”, Developments in Sedimentology, 30B, Elsevier: Amsterdam, 2 Vols.

American Waterworks Association, (1990). Water Quality and Treatment: A Handbook of Community Water Supplies. McGraw-Hill Inc., NY, 1194 pp.

Andreae, M. O., & Froelich, P. N., (1984). “Arsenic, Antimony and Germanium Biogeochemistry in the Baltic Sea”, Tellus, 36B, 101-117.

Archer, D., Emerson, S., & Smith, C.R., (1989). “Direct Measurement of the Diffusive Sub-layer at the Deep Sea Floor using Oxygen Micro-electrodes,” Nature, 340, 623-626.

Asmund, G., (1992). “Pollution from the Marine Tailings Disposal at the Lead-Zinc Mine at Maarmorilik, West Greenland”, In Mining in the Arctic, Balkema: Rotterdam.

Asmund, G., Broman, P.G., & Lindgren, G., (1994). “Managing the Environment at the Black Angel Mine, Greenland”, Int. J. Sfc. Min. Recl. Env., 8, 37-40.

Atkins, R.J., (1993). Sediment Suspension under Irregular "Groupy" Waves: A Laboratory Experiment. Unpublished M.Sc.Thesis, University of Toronto: Toronto, 219 pp.

Bell, A. V., (1987). “Prevention of Acid Generation in Base Metal Tailings and Waste Rock”, Proceedings of Acid Mine Drainage Seminar/Workshop, Halifax, Nova Scotia, March 23-26.

Belzile, N., (1988). “The Fate of Arsenic in Sediments of the Laurentian Trough”, Geochim. Cosmochim. Acta., 52, 2293-2302.

Brander, R.W., (1991). Bedform Constraints on Sediment Re-suspension and Transport under Shoaling and Breaking Waves. Unpublished M.Sc. Thesis, University of Toronto, 182 pp.

Bruland, K.W. (1989) Complexation of zinc by natural organic ligands in the Central North Pacific. Limnol. Oceanogr., 34, 267-283.

Bruland, K.W. (1992) Complexation of cadmium by natural organic ligands in the Central North Pacific. Limnol. Oceanogr., 37, 1008-1017.

Calcagno, T.H. and G.M. Ashley, (1991). “Sedimentation Processes in an Impoundment, Union Lake, New Jersey”, Environmental Geological Water Science, 6(4), 237-246.

Carter, R.W.G., (1991). Coastal Environments, Academic Press: London, 617 pp.

Cullen, W. R., & Reimer, K. J. (1989). “Arsenic Speciation in the Environment”, Chem. Rev., 89, 713-764.

Dave N.K., Lim T.P., Horne D., Boucher Y., and Stuparyk R. (1997). “Water cover on reactive tailings and wasterock: Laboratory studies of oxidation and metal release characteristics”, Proceedings, Fourth International, Conference on Acid Rock Drainage (eds. K. Ferguson *et al.*). Vancouver, B.C., May 31 - June 6, 1997.

David, D. J., & Nicholson, R. V., (1995). “Field Measurements for Determining Rates of Sulphide Oxidation”, Sudbury '95 Conference on Mining and the Environment, Sudbury, Ontario.

Davies, A.G. and R.H. Wilkinson, (1979). “Sediment Motion Caused by Surface Water Waves”, Proceedings 16th Coastal Engineering Conference, ASCE: New York, 1577-1595.

Davison, W., (1993). “Iron and Manganese in Lakes”, Earth Sci. Rev., 34, 119-163.

Demopoulos, G.P., & Papangelakis, V.G. (1989). “Recent Advances in Refractory Gold Processing”, CIM, 82 (931), 85-91.

Dinardo, O., *et al.*, (1991). “Study on Metals Recovery/Recycling from Acid Mine Drainage”, Phase 1a: Literature Survey, MEND Treatment Committee, 1-41.

Dingler, J.R., (1979). “The Threshold of Grain Motion under Oscillatory Flow in a Laboratory Channel”, Journal of Sedimentary Petrology, 49, 287-294.

Dyer, K.R., (1986). Coastal and Estuarine Sediment Dynamics, John Wiley and Sons: Chichester, 342 pp.

Ellis, D.V., Pedersen, T.F., Poling, G.W., Pelletier, C.A., & Horne, I. (1995). “A review of 23 years of STD: Island Copper Mine, Canada”. Marine Georesources and Geotechnology, 13, 59-99.

Equity Silver Mines Limited, (1992). Closure Plan - January 1992, 9 pp plus Appendices.

Evangelou, V. P., & Zhang, Y. L., (1995). “A Review: Pyrite Oxidation Mechanisms and Acid Mine Drainage Prevention”. Crit. Rev. Environ. Sci. Technol, 25(2), 141-199.

Gorham, E., and F. M. Boyce, (1989). “Influence of Lake Surface Area and Depth upon Thermal Stratification and the Depth of the Summer Thermocline”, Journal of Great Lakes Research, 15(2):233 - 245.

Gould, W.D., Bechard, G., & Lortie, L. (1994). The Nature and Role of Microorganisms in the Tailings Environment. In J.L. Jambor & D.W. Blowes (Eds.), Short Course Handbook on Environmental Geochemistry of Sulphide Mine-Wastes. (pp. 185-199). Waterloo, Ontario: Mineral. Assoc. of Can.

Gibbs, R.J., *et al*., (1971). “The Relationship between Sphere Size and Settling Velocity”, Journal of Sedimentary Petrology, 41(1), 7-18.

Gundersen, J. K., & Jorgensen, B. B., (1990). “Microstructure of Diffusive Boundary Layers and the Oxygen Uptake of the Sea Floor”, Nature, 345, 604-607.

Haque, K. E., (1992). “Amendments for the Control of Acid Mine Drainage”, CANMET Report, Project 30.14.01, 1-14.

Harris, G. B., & Monette, S., (1988). “The Stability of Arsenic-Bearing Residues”, Arsenic Metallurgy Fundamentals and Applications, The Metallurgical Society:Warrendale, pp. 469-488.

Heltzel, S.B. and A.M. Teeter, (1987). “Settling of Cohesive Sediments”, Coastal Sediments '87, ASCE: New York, 63-70.

Higgs, T.W. (1993). Technical guide for the environmental management of cyanide in mining. Mining Association of British Columbia.

Huerta-Diaz, M. A., & Morse, J. W., (1990). “A Quantitative Method for Determination of Trace Metal Concentrations in Sedimentary Pyrite”, Mar. Chem., 29(2-3), 119-144.

Jambor, A. J. L., (1994). “Mineralogy of Sulfide-Rich Tailings and Their Oxidation Products”, The Environmental Geochemistry of Sulfide Mine-Wastes, 22, 59-102.

Jaynes, D.B.A., Rogowski, A.S., & Pionke, H.B. (1984). Acid mine drainage from reclaimed coal strip mines, 2., Simulation results of model. Wat. Resour. Res., 20(2), 243-250.

Johansen, P., Hanse, M.M., Asmund, G., & Nielsen, P.B. (1991). “Marine organisms as indicators of heavy pollution – Experience from 16 years of monitoring at lead-zinc mine in Greenland”, Chemistry and Ecology, 5, 35-55.

Jorgensen, B. B., & Des Marais, D. J., (1990). “The Diffusive Boundary Layer of Sediments: Oxygen Microgradients over a Microbial Mat”, Limnol. Oceanogr., 35, 1343-1355.

Jorgensen, B. B., & Revsbech, N. P., (1985). “Diffusive Boundary Layers and the Oxygen Uptake of Sediments and Detritus”, Limnol. Oceanogr., 30, 111-122.

Kamphuis, J. W., (1975). The Coastal Mobile Bed Model, Civil Engineering Research Report No.

75, Queen’s University: Kingston

Klinkhammer, G., Heggie, D.T., & Grham, D.W., (1982). “Metal Diagenesis in Oxic Marine Sediments”, Earth and Planetary Science Letters, 61, 211-219.

Komar, P.D. and M.C. Miller, (1975a). "Sediment Threshold under Oscillatory Waves”, Proceedings, 14th Coastal Engineering Conference, ASCE: New York, 756-775.

Komar P.D. and M.C. Miller, (1975b). “On the Comparison between the Threshold of Motion under Waves and Unidirectional Currents with a Discussion of the Practical Evaluation of the Threshold”, Journal of Sedimentary Petrology, 45, 362-367.

Komar, P.D. and M.C. Miller, (1973). “The Threshold of Sediment Movement under Oscillatory Water Waves”, Journal of Sedimentary Petrology, 43, 1101-1110.

Krause, E., & Ettel, V. A., (1985). “Ferric Arsenate Compounds: Are They Environmentally Safe? Solubilities of Basic Ferric Arsenates”, Proc. of Impurity Control and Disposal, CIM Ann. Conf., Vancouver, B.C.

Krause, E., & Ettel, V. A., (1989). “Solubilities and Stabilities of Ferric Arsenate Compounds”, Hydrometallurgy, 22, 311-337.

Kwong, J. Y., (1993). Prediction and Prevention of Acid Rock Drainage from a Geological and Mineralogical Perspective, MEND Project 1.32.1.

Lawrence, G.A., *et al*., (1991). “Wind-Wave-Induced Suspension of Mine Tailings in Disposal Ponds - A Case Study”, Canadian Journal of Civil Engineering, 18, 1047-1053.

Li M.G., Aube B., and St.-Arnaud L., (1997). “Considerations in the use of shallow water covers for decommissioning reative tailings.”, Proceedings, Fourth International Conference on Acid Rock Drainage (eds. K. Ferguson *et al.*), Vancouver, B.C., May 31-June 6, 1997.

Lindberg, S.E., & Harriss, R.C. (1974). “Mercury-organic matter association in estuarine sediments and interstitial water”. Environ. Sci. Tech., 8 (5), 459-462.

Lockwood, R.A., & Chen, K.Y. (1973). “Adsorption of HG(II) by Hydrous Manganese Oxide”. Environmental Science and Technology, 7 (11), 1028-1034.

Luther III, G. W., (1987). “Pyrite Oxidation and Reduction: Molecular Orbital Theory Considerations”, Geochim. Cosmochim. Acta, 51, 3193-3199.

Martin, A., (1996). The Geochemistry of Fe, Mn, Ni, Cu, Zn and As in the Water Column, Sediments and Pore-waters in a Seasonally Anoxic Lake. Masters Thesis, UBC.

McKee, J.D., Wilson, T.P., & Long, D.T., (1989). “Pore Water Profiles and Early Diagenesis of Mn, Cu and Pb in Sediments from Large Lakes”, J. Great Lakes Res., 5, 68-83.

Middelburg, J. J., (1989). “A Simple Rate Model for Organic Matter Decomposition in Marine Sediments”, 53, 1577-1581.

Morel, F. M. M., & Hudson, R. J. M., (1985). “The Geobiolobical Cycle of Trace Elements in Aquatic Systems: Redfield Revisited”, Chemical Processes in Lakes, Wiley-Interscience.

Morfett, K., Davison, W., & Hamilton-Taylor, J., (1988). “Trace Metal Dynamics in a Seasonally Anoxic Lake”. Environ. Geol. Wat. Sci., 11, 107-114.

Morin, K.A. (1993). “Rates of sulphide oxidation in submerged environments: implications for subaqueous disposal”. In 17th Annual Mine Reclamation Symposium, Port Hardy, British Columbia.

Moses, C. O., & Herman, J. S., (1991). “Pyrite Oxidation at Circumneutral pH”. Geochim. Cosmochim. Acta, 55, 471-482.

Moses, C. O., Nordstrom, D.K., Herman, J.S., & Mills, A.L., (1987). “Aqueous Pyrite Oxidation by Dissolved Oxygen and by Ferric Iron", Geochim. Cosmochim. Acta, 51, 1561-1571.

Murray, S.P., (1970). "Settling Velocities and Vertical Diffusion of Particles in Turbulent Water", Journal of Geophysical Research, 75(9), 1647-1654.

Mustin, C., Berthelin, J., Marion, P., & de Donato, P. (1992) Corrosionand electrochemical oxidation of a pyrite by *Thiobaccillus ferrooxidans.* Appl. Environ. Microbiol.,58, 1175.

Nicholson, R. V., *et al*., (1995). "A New Oxygen Technique to Provide Rapid Assessment of Tailings Reactivity in the Field and the Laboratory", Sudbury '95 Conference on Mining and the Environment, Sudbury, Ontario.

Nicholson, R.V. (1994) "Iron-sulphide oxidation mechanisms: Laboratory studies". In J.L. Jambor & D.W. Blowes (Eds.), Short Course Handbook on Environmental Geochemistry of Sulphide Mine-Wastes. (pp. 163-183). Waterloo, Ontario: Mineral Assoc. of Can.

Nicholson, R. V., Gillham, R.W., & Reardon, E.J., (1990). "Pyrite Oxidation in Carbonate-Buffered Solution: 2. Rate Control by Oxide Coatings", Geochim. Cosmochim. Acta, 54, 395-402.

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 Holberg Inlets, B.C.", Can. J. Earth. Sci., 22, 1474-1484.

Pedersen, T. F., & Losher, A. J. (Ed.)., (1988). Diagenetic Processes in Aquatic Mine Tailings Deposits in British Columbia. Springer-Verlag: Berlin.

Pedersen, T.F., Mueller, B., McNee, J.J., & Pelletier. (1991). "Diagnetic Reactivity of Mine Tailings in Mesotrophic and Oligotrophic Lakes in Manitoba and British Columbia". Second International Conference on the Abatement of Acid Drainage. Montreal, Quebec.

Pedersen, T.F., McNee, J.J., Mueller, B., Flather, D.H., & Pelletier, C.A. (1994). "Geochemistry of Reactive Tailings in Anderson Lake, Manitoba, and Buttle Lake, British Columbia". Third International Conference on the Abatement of Acid Drainage, Pittsburgh, PA.

Philpott Associates, (1991). Equity Silver Mines: Tailings Mobilization Study.

Reimer, K. J., & Thompson, J. A. J., (1988). "Arsenic Speciation in Marine Interstitial Water: The Occurrence of Organoarsenicals" Biogeochemistry, 6, 211-237.

Reimers, C. E., Kalhorn, S., Emerson, S.R., & Nealson, K.H., (1984). "Oxygen Consumption Rates in Pelagic Sediments from the Central Pacific: First Estimates from Microelectrode Profiles", Geochim. Cosmochim. Acta., 48, 903-910.

Reimers, C. E., & Smith, K. L., Jr., (1986). "Reconciling Measured and Predicted Fluxes of Oxygen Across the Deep Sea Sediment-Water Interface", Limnol. Oceanogr., 31(2), 305-318.

Revsbech, N.P., Sorensen, J., Blackburn, T.H. & Lomholt, J.P. (1980). "Distribution of oxygen in marin sediments measured with micro-electrodes". *Limnol. Oceanogr*., 25, 403-411.

Revsbech, N. P., Madsen, B., & Jorgensen B.B., (1986). "Oxygen Production and Consumption in Sediments Determined at High Spacial Resolution by Computer Simulation of Oxygen Microelectrode Data", Limnol. Oceanogr., 31, 293-304.

Ritcey, G. M., (1989). Tailings Management: Problems and Solutions in the Mining Industry. Elsevier.

Sahami, A. R., *et al*., (1996). "The SeaOxy Profiler: A New Autonomous Instrument for the Remote Measurement of Sediment Oxygen Demand in Environmental Studies", Mar. Chem., (in press).

Sawlan, J. J., & Murray, J. W., (1983). "Trace Metal Remobilization in the Interstitial Waters of

Red Clay and Hemipelagic Marine Sediments", Earth. Planet. Sci. Lett., 64, 213-230.

Singer, P. C., & Stumm, W., (1970). Acid Mine Drainage: The rate determining step.", Science, 167, 1121-1123.

Smith, A., & Mudder, T. (1991). Chemistry and Treatment of Cyanidation Wastes. London: Mining Journal Books Ltd.

Soulsby, R.L., (1983). "The Bottom Boundary Layer of Shelf Seas." Chapter 5 in Physical Oceanography of Coastal and Shelf Seas, B. Johns (Ed.), Oceanography Series 35, Elsevier: Amsterdam, 189-266.

Steffen Robertson and Kirsten (B. C.) Inc., *et al*., (1990). Draft Acid Rock Drainage Technical Guide. A BCAMD Task Force Report.

Sternberg, R.W., (1971). "Measurements of Incipient Motion of Sediment Particles in the Marine Environment", Marine Geology, 10, 113-119.

Sternberg, R.W., (1972). "Predicting Initial Motion and Bedload Transport of Sediment Particles in the Shallow Marine Environment", Shelf Sediment Transport, Process and Pattern, Dowden Hutchinson Ross: Stroudsburg, PA, 61-83.

Stumm, W., & Morgan, J. J., (1981). Aquatic Chemistry, 2nd Ed., John Wiley & Sons: New York.

Sweerts, J.-P. R., *et al*., (1991). "Oxygen-Consuming Processes at the Profundal and Littoral Sediment-Water Interface of a Small Meso-Eutrophic Lake (Lake Vechten, The Netherlands)", Limnol. Oceanogr., 36, 1124-1133.

Turner, R. J. W., (1987). "The Genesis of Stratiform Lead-Zinc Deposits, Jason Property, Macmillan Pass, Yukon", Thesis., (Publisher *etc.*) (pp. 239).

U.S. Army Coastal Engineering Research Centre, (1984). Shore Protection Manual, Vicksburg, MS.

Vachon, D., *et al*., (1987). "Treatment of Acid Mine Water and the Disposal of Lime Neutralization Sludge", Proceedings of Acid Mine Drainage Seminar/Workshop, Halifax, Nova Scotia, March 23-26, 537-564.

Westerlund, S. F. G., *et al*., (1986). "Benthic Fluxes of Cadmium, Copper, Nickel, Zinc and Lead in the Coastal Environment", Geochim. Cosmochim. Acta, 50, 1289-1296.

Wills, B.A. (1992). Mineral Processing Technology (5th ed.). Pergamon Press.

E3.0 ENGINEERING STANDARDS

CSA (1991). “Canadian Standards Association, Risk Analysis Requirements and Guidelines.” Quality Management, A National Standard of Canada. 42pp.

Fell, R., (1994). “Landslide Risk Assessment and Acceptable Risk.” Canadian Geotechnical Journal, V. 31, p. 261.

ICOLD (1994). “Embankment Dams Granular Filters and Drains.” Review and Recommendations. Bulletin 95. 255 pp.

ICOLD (1995a). “International Commission of Large Dams.” Tailings Dams and Seismicity, Bulletin 98, 60 pp.

ICOLD (1996). “International commission of Large Dams.” Dam Failures Statistical Analysis. Bulletin 99, 73 pp.

Ripley, C.F., (1988). “Comments on Q.61 - R 29 and R 55.” Proceedings of XVI ICOLD. Vol. V.

USCOLD (1994). “United States Committee on Large Dams.” Tailings Dam Incidents. November 1994.

ARD PREDICTION METHODOLOGY

Excerpt from

The Draft Acid Rock Drainage Technical Guide - Vol. 1

British Columbia Acid Mine Drainage Task Force Report
August 1989