

Series G: General Guidelines



G4

BEST PRACTICE
GUIDELINE

Impact Prediction

Best Practice Guidelines for Water Resource Protection in the South African Mining Industry

DIRECTORATE: RESOURCE PROTECTION & WASTE



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Consultants:

Pulles Howard & de Lange Inc.
P O Box 861
AUCKLAND PARK
2006
Republic of South Africa

Golder Associates Africa
PO Box 6001
HALFWAY HOUSE
1685
Republic of South Africa

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This document is the fourth in a series of the following general aspects Best Practice Guideline documents:

BPG G1: Storm Water Management

BPG G2: Water and Salt Balances

BPG G3: Water Monitoring Systems

BPG G4: Impact Prediction

BPG G5: Water Management Aspects for Mine Closure

ACKNOWLEDGE- MENTS

Authors

Mr William Pulles (Golder Associates)

Specialists

Dr Brent Usher (University of Free State)

Mr Nico Bezuidenhout (Golder Associates)

Mrs Riana Munnik (DWAF)

Mr Tefo Tshabidi (DWAF)

Since 1999 a number of steering committee meetings and stakeholder workshops were held at various stages of the development and drafting of this series of Best Practice Guidelines for Water Resource Protection in the South African Mining Industry.

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APPROVALS

This document is approved by the Department of Water Affairs and Forestry



RIANA MUNNIK

Deputy Director: Resource Protection and Waste: Mines

Date: 10/12/2008



NNDANGANENI MUSEKENE

Director: Resource Protection and Waste

Date: 2008/12/22



DEBORAH MOCHOTLHI

Chief Director: Water Use

Date: 19/01/2009

PREFACE

Water is typically the prime environmental medium (besides air) that is affected by mining activities. Mining adversely affects water quality and poses a significant risk to South Africa's water resources. Mining operations can further substantially alter the hydrological and topographical characteristics of the mining areas and subsequently affect the surface runoff, soil moisture, evapo-transpiration and groundwater behaviour. Failure to manage impacts on water resources (surface and groundwater) in an acceptable manner throughout the life-of-mine and post-closure, on both a local and regional scale, will result in the mining industry finding it increasingly difficult to obtain community and government support for existing and future projects. Consequently, sound management practices to prevent or minimise water pollution are fundamental for mining operations to be sustainable.

Pro-active management of environmental impacts is required from the outset of mining activities. Internationally, principles of sustainable environmental management have developed rapidly in the past few years. Locally the Department of Water Affairs and Forestry (DWAF) and the mining industry have made major strides together in developing principles and approaches for the effective management of water within the industry. This has largely been achieved through the establishment of joint structures where problems have been discussed and addressed through co-operation.

The Bill of Rights in the Constitution of the Republic of South Africa, 1996 (Act 108 of 1996) enshrines the concept of sustainability; specifying rights regarding the environment, water, access to information and just administrative action. These rights and other requirements are further legislated through the National Water Act (NWA), 1998 (Act 36 of 1998). The latter is the primary statute providing the legal basis for water management in South Africa and has to ensure ecological integrity, economic growth and social equity when managing and using water. Use of water for mining and related activities is also regulated through regulations that were updated after the promulgation of the NWA (Government Notice No. GN704 dated 4 June 1999).

The NWA introduced the concept of Integrated Water Resource Management (IWRM), comprising all aspects of the water resource, including water quality, water quantity and the aquatic ecosystem quality (quality of the aquatic biota and in-stream and riparian habitat). The IWRM approach provides for both resource directed and source directed measures. Resource directed measures aim to protect and manage the receiving environment. Examples of resource directed actions are the formulation of resource quality objectives and the development of associated strategies to ensure ongoing attainment of these objectives; catchment management strategies and the establishment of catchment management agencies (CMAs) to implement these strategies.

On the other hand, source directed measures aim to control the impacts at source through the identification and implementation of pollution prevention, water reuse and water treatment mechanisms.

The integration of resource and source directed measures forms the basis of the **hierarchy of decision-taking** aimed at protecting the resource from waste impacts. This hierarchy is based on a *precautionary approach* and the following order of priority for mine water and waste management decisions and/or actions is applicable:

RESOURCE PROTECTION AND WASTE MANAGEMENT HIERARCHY



The documentation describing **Water Resource Protection and Waste Management** in South Africa is being developed at a number of different levels, as

described and illustrated in the schematic diagram on this page.

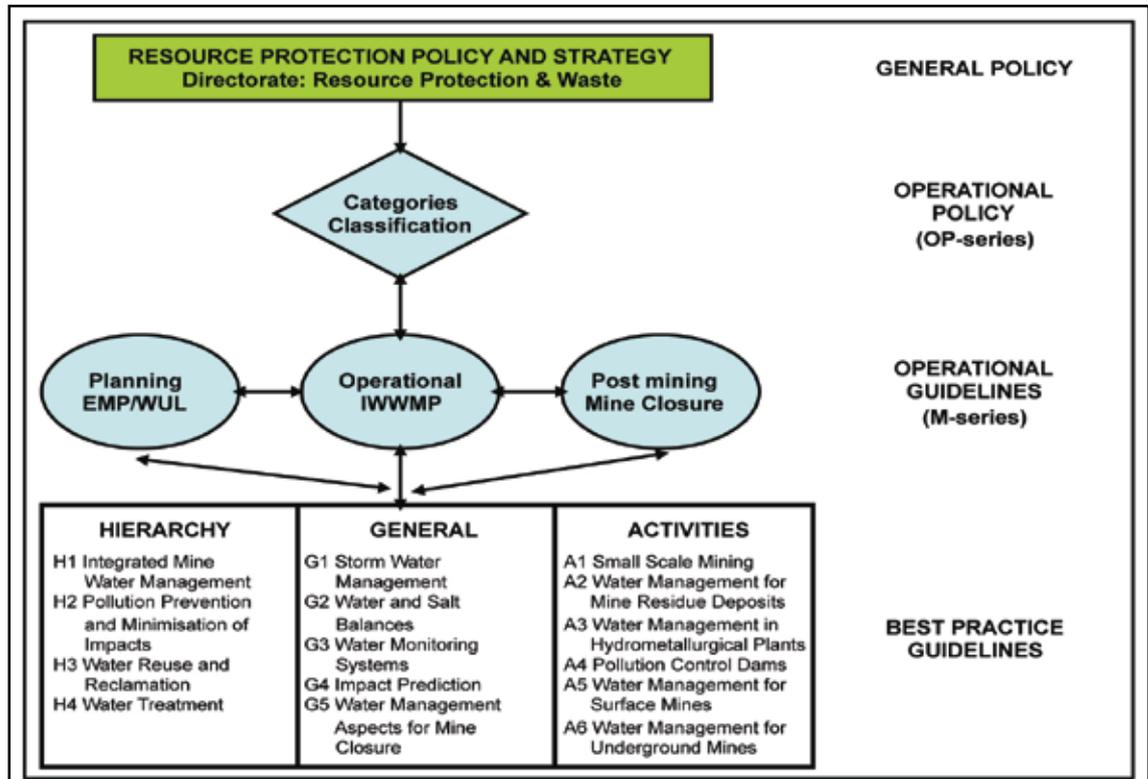
The overall Resource Protection and Waste Management Policy sets out the interpretation of policy and legal principles as well as functional and organisational arrangements for resource protection and waste management in South Africa.

Operational policies describe the rules applicable to different categories and aspects relating to waste discharge and disposal activities. Such activities from the mining sector are categorised and classified based on their potential risks to the water environment.

Operational Guidelines contain the requirements for specific documents e.g. licence application reports.

Best Practice Guidelines (BPG's) define and document best practices for water and waste management.

Schematic Diagram of the Mining Sector Resource Protection and Waste Management Strategy



The DWAF has developed a series of **Best Practice Guidelines** (BPGs) for mines in line with International Principles and Approaches towards sustainability. The series of BPGs have been grouped as outlined below:

BEST PRACTICE GUIDELINES dealing with aspects of DWAF's water management **HIERARCHY** are prefaced with the letter **H**. The topics that are covered in these guidelines include:

- H1. Integrated Mine Water Management
- H2. Pollution Prevention and Minimisation of Impacts
- H3. Water Reuse and Reclamation
- H4. Water Treatment

BEST PRACTICE GUIDELINES dealing with **GENERAL** water management strategies, techniques and tools, which could be applied cross-sectoral and always prefaced by the letter **G**. The topics that are covered in these guidelines include:

- G1. Storm Water Management
- G2. Water and Salt Balances
- G3. Water Monitoring Systems
- G4. Impact Prediction
- G5. Water Management Aspects for Mine Closure

BEST PRACTICE GUIDELINES dealing with specific mining **ACTIVITIES** or **ASPECTS** and always prefaced by the letter **A**. These guidelines address the prevention and management of impacts from:

- A1. Small-Scale Mining
- A2. Water Management for Mine Residue Deposits
- A3. Water Management in Hydrometallurgical Plants
- A4. Pollution Control Dams
- A5. Water Management for Surface Mines
- A6. Water Management for Underground Mines

The development of the guidelines is an inclusive consultative process that incorporates the input from a wide range of experts, including specialists within and outside the mining industry and government. The process of identifying which BPGs to prepare, who should participate in the preparation and consultative processes, and the approval of the BPGs was managed by a Project Steering Committee (PSC) with representation by key role-players.

The BPGs will perform the following functions within the hierarchy of decision making:

- Utilisation by the mining sector as input for compiling water use licence applications (and other legally required documents such as EMPs, EIAs, closure plans, etc.) and for drafting licence conditions.
- Serve as a uniform basis for negotiations through the licensing process prescribed by the NWA.
- Used specifically by DWAF personnel as a basis for negotiation with the mining industry, and likewise by the mining industry as a guideline as to what the DWAF considers as best practice in resource protection and waste management.
- Inform Interested and Affected Parties on good practice at mines.

The information contained in the BPGs will be transferred through a structured knowledge transfer process, which includes the following steps:

- Workshops in key mining regions open to all interested parties, including representatives from the mining industry, government and the public.
- Provision of material to mining industry training groups for inclusion into standard employee training programmes.
- Provision of material to tertiary education institutions for inclusion into existing training programmes.
- Provision of electronic BPGs on the DWAF Internet web page.

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ABBREVIATIONS

Accounting	
AP	Acid Potential
ARD	Acid Rock Drainage
ASTM	American Society for Testing and Materials
BPG	Best Practice Guideline (documents in this series)
CANMET	Canada Centre for Mineral and Energy Technology
DTM	Digital Terrain Model
DWAF	Department of Water Affairs and Forestry
EC	Electrical Conductivity
ECA	Environmental Conservation Act, 1989 (Act 73 of 1989)
EIA	Environmental Impact Assessment
EMP	Environmental Management Programme
EPMA	Electron Probe Microanalysis
EPRI	Electrical Power Research Institute
GIS	Geographic Information System
GN704	Government Notice 704 promulgated in terms of NWA on 4 June 1999
GPS	Global Positioning System
ICPMS	Inductively Coupled Plasma Mass Spectroscopy
IWM	Integrated water management
IWMP	Integrated water management plan
MPRDA	Minerals and Petroleum Resources Development Act (Act 28 of 2002)
NEMA	National Environmental Management Act, 1998 (Act 107 of 1998)
NP	Neutralising Potential
NNP	Net Neutralising Potential = NP – AP
NPR	Neutralising Potential Ratio = NP/AP
NWA	National Water Act, 1998 (Act 36 of 1998)
PAG	Potentially ARD Generating

OCM	Oxygen Consumption Method
QA/QC	Quality Assurance / Quality Control
RDF	Residue Disposal Facility
SEM	Scanning Electron Microscopy
TDF	Tailings Disposal Facility
TDS	Total Dissolved Solids
UNEP	United Nations Environment Programme
USEPA	United States Environmental Protection Agency
WASP	Waste Aquifer Separation Principle
WRC	Water Research Commission
WULA	Water Use Licence Application
XRD	X-Ray Diffraction
XRF	X-Ray Fluorescence



GLOSSARY

In assessing the definitions given below, it must be understood that the definitions as provided in the NWA and Government Notice 704 (GN704) are primary.

Active management system: A management system that may require external energy inputs (such as electrical power) or continuous operator attention for its continued successful operation.

Aquifer: a geological formation which has structures or textures that hold water or permit appreciable water movement through them

Audit: A systematic, documented, periodic and objective evaluation of how well management systems and equipment are performing, with the aim of facilitating management control of practices and to assess compliance with relevant policies and objectives, which include meeting regulatory requirements.

Catchment: In relation to a watercourse or watercourses or part of a watercourse, means the area from which any rainfall will drain into the watercourse or watercourses or part of a watercourse, through surface flow to a common point or common points. (National Water Act, 1998 (Act 36 of 1998)).

Category A Mines: Those mines that exploit orebodies that are associated with sulphide minerals or any other reactive minerals, either in the ore, overburden or waste material.

Clean water: Water that has not been affected by pollution.

Dirty water: Water that contains waste.

Groundwater: Water that occurs in the voids of saturated rock and soil material beneath the ground surface is referred to as groundwater and the body within which the groundwater is found is referred to as an aquifer.

Life cycle costing: Life-cycle costing refers to the process whereby all costs associated with the system (e.g. capital cost, operational cost, maintenance costs, closure and rehabilitation cost, impact mitigation costs, etc.) as applied to the defined life cycle are considered

Life of mine: The life of mine includes all the phases of the mine's existence from the conceptual and planning phases, through design, construction, operation and decommissioning to the post-closure and aftercare phases.

Mitigation: Measures taken to reduce adverse impacts on the environment.

Passive management system: A management system that does not require external energy inputs (such as electrical power) or continuous operator attention for its continued successful operation.

Pollution: Pollution means the direct or indirect alteration of physical, chemical or biological properties of a water resource so as to make it –

- (a) less fit for any beneficial purpose for which it may reasonably be expected to be used;
or
- (b) harmful or potentially harmful –
 - (aa) to the welfare, health or safety of human beings;
 - (bb) to any aquatic or non-aquatic organisms;
 - (cc) to the resource quality; or

(dd) to property.

(National Water Act, 1998 (Act 36 of 1998))

Precautionary principle: This refers to the principle that in the absence of actual data to demonstrate an alternative conclusion, the most conservative assumption will be made and precautionary management measures will need to be applied.

Prediction Specialist: a person with the requisite training, skill and expertise to participate in an impact prediction exercise and who is capable of signing a declaration of his expertise and ability to undertake the work in question and his/her willingness to subject him/herself to independent specialist review.

Prevention: Measures taken to minimize the release of contaminants to the environment.

Residue: Residue includes any debris, discard, tailings, slimes, screenings, slurry, waste rock, foundry sand, beneficiation plant waste, ash and other waste product derived from or incidental to the operation of a mine or activity and which is stockpiled, stored or accumulated for potential reuse or recycling or which is disposed of. (Government Notice 704 of 4 June 1999.)

Residue deposits: Residue deposits include any dump, tailings dams, slimes dams, ash dump, waste rock dump, in-pit deposit and any other heap, pile or accumulation of residue. (Government Notice 704 of 4 June 1999.)

Resource quality: means the quality of all the aspects of a water resource including (National Water Act, 1998 (Act 36 of 1998))-

- (a) the quantity, pattern, timing, water level and assurance of instream flow;
- (b) the water quality, including the physical, chemical and biological characteristics of the water;
- (c) the character and condition of the instream and riparian habitat; and
- (d) the characteristics, condition and distribution of the aquatic biota

Risk assessment: The qualitative and quantitative evaluation performed in an effort to define the risk posed to human health or the environment by the presence or potential presence and use of specific pollutants.

Seepage: The act or process involving the slow movement of water or another fluid through a porous material like soil, slimes or discard.

Siting: The process of choosing a location for a facility.

Slope: Slope is a dimensionless number and is defined by the vertical distance (drop) divided by the horizontal distance.

Suitably qualified and experienced person: Suitably qualified means a person having a level of training and experience with the type of work to be done and recognised skills in the type of work to be done.

Surface water: All water naturally open to the atmosphere (rivers, lakes, reservoirs, streams, impoundments, seas, estuaries, etc.); also refers to springs, wells, or other collectors that are directly influenced by surface water.

Watercourse: Watercourse means –

- a) a river or spring;
- b) a natural channel in which water flows regularly or intermittently;
- c) a wetland, lake or dam into which, or from which, water flows; and

any collection of water which the Minister may, by notice in the Gazette, declare to be a watercourse, and a reference to a watercourse includes, where relevant, its beds and banks. (National Water Act, 1998 (Act 36 of 1998)).

Water resource: Includes a watercourse, surface water, estuary, or aquifer. (National Water Act, 1998 (Act 36 of 1998))

Water system: Water system includes any dam, any other form of impoundment, canal, works, pipeline and any other structure or facility constructed for the retention or conveyance of water. (Government Notice 704 of 4 June 1999.)

1

INTRODUCTION AND OBJECTIVES OF THIS BEST PRACTICE GUIDELINE

1.1 INTRODUCTION

The need to make predictions of future impact of mining operations on the water resource is fundamental to the discipline of environmental management at mines and is a requirement of environmental impact assessments (EIA), water use licence applications (WULA), environmental management programmes (EMP), mine closure plans and others. In each instance there is a need to understand the future impact of a current activity and to then determine whether the management measures applied to that current activity are appropriate or whether they should be modified.

The need for future predictions of impact on the water resource is particularly important at all mines that mine ore bodies or produce mine wastes that contain sulphide minerals as these will oxidise over time, releasing sulphates and other contaminants. Examples of such mines include coal, gold, base metals, platinum, diamonds and others. Other reactive minerals that release contaminants through other processes such as dissolution or leaching also require predictions of future behaviour and impact.

Various tools and techniques have been developed to enable the assessment of future water quality impacts from mining operations to be made. Most of these tools have been developed in North America, with significant developments also being made in Australia and Europe. It is important to understand that these tools were generally developed to answer very specific questions that are relevant to the regulatory environment in those countries, potentially limiting their utility in South Africa. A prime example of this limitation is the acid base accounting (ABA) technique which is a test intended to determine the risk that a particular material might generate acidity at some point in future. The test is not designed to answer any questions relating to how acid the material might become, when it would become acidic, for how long acidic conditions would continue and what other contaminants might be released when the material goes acid or in the event that it does not turn acid.

Historically, the mining regulatory environment in North America has focused on the question of whether acidic conditions will form and what metals will be liberated in the event of the onset of acidic conditions and thus developed the ABA technique. In South Africa, where water is much scarcer and a high degree of water recycling is applied, a number of additional water quality aspects are pertinent – most notably salinity and sulphates and it is therefore important that predictive techniques applied are able to provide information on a wide range of contaminants, i.e. more than just acidity as obtained from ABA. In the particular situation of predictions made as part of a mine closure planning process, where the granting of mine closure implies that the State assumes long-term liability and risk for water quality impacts, it is necessary to not only know the risk that a particular event may occur (e.g. drainage turns acidic) but also when it will happen, how severe it will be and how long it will persist in order that financial liability to manage the impact can be determined.

For the above reasons caution should be exercised in simply applying impact prediction approaches and protocols developed to suit the regulatory environment in other countries, to South Africa. The purpose and objective of this BPG is therefore to present the impact prediction approach that is applicable to the South African regulatory environment with regard to water resource management.

The process of making predictions of potential impact on the water resource at some future time and the reliability of these predictions is dependant on the following elements:

- 1) Correct formulation of the key questions that need to be answered by the impact prediction exercise within the context of South African legislation and regulations.

- 2) Characterization and knowledge of the sources that give rise to the impact.
- 3) Characterization and knowledge of the environmental pathway along which the impact migrates.
- 4) Characterization and knowledge of the receptor that experiences the impact.
- 5) Knowledge of the capabilities and limitations of the various approaches and tools that can be used to make impact predictions.
- 6) Understanding of the issues of risk and uncertainty inherent in the tools and the assessment procedure and how these combine to affect the confidence that can be placed in the impact prediction that is made.
- 7) Access to the various tools and proper training and experience in selecting the appropriate tools for the assessment being undertaken and in using the selected tools.
- 8) Adequate and appropriate independent review of the prediction methodology and results.
- 9) Design and implementation of an appropriate post-prediction monitoring programme, to support the validation and calibration of the predictions.

These elements will be addressed in this BPG.

1.2 OBJECTIVES OF THIS BPG

This BPG, due to the nature of the material covered and due to the general scarcity of specialist knowledge on the topics covered, needs to strike a balance between providing information and guidance on the one hand and making very specific recommendations on the other. The need to make a number of specific recommendations is also linked to the fact that the outcome of impact prediction is very often the development of a mine closure application which seeks to transfer liability and risk from the mine owner to the State.

The objectives of this BPG are:

- 1) To present the impact assessment and prediction framework and methodology based on risk assessment principles.
- 2) To provide specific guidance on the types of questions that need to be considered and answered when undertaking impact predictions and evaluating management options for different mining scenarios at different stages in the mine life cycle, in order that appropriate decisions can be made.

- 3) To give a brief overview of physical, chemical and biological processes involved in generation of acidic, neutral and saline drainages, as well as groundwater flow and contaminant migration, in order to give users of the BPG a basic understanding of the issues that affect the impact prediction.
- 4) To describe the capabilities and limitations of different approaches, tools and techniques available for impact prediction in order that users of this BPG can select and integrate the appropriate tools for each type of assessment.
- 5) To describe the elements of uncertainty that are inherent in any impact prediction, given the nature of the process (source term, pathway and receptor) being evaluated, the assumptions and uncertainty inherent in a data collection exercise and the limitations and assumptions inherent in prediction tools.
- 6) To provide guidance on the nature of a monitoring, predictive model validation and calibration programme and independent review necessary to give the required confidence in the impact prediction.
- 7) To provide guidance on the contents of a water resource impact prediction report.
- 8) To provide capacity building for DWAF officials in the review and understanding of impact prediction exercises.

1.3 APPLICABILITY, STRUCTURE AND FOCUS OF THIS BPG

As impact prediction is fundamental to assessment and management of the impacts of mining on the water resource, this BPG is applicable to all mining operations, regardless of whether they are in the exploration, planning, operational or closure phase. All mines have a potential to have an impact on the water resource and the objective of an impact prediction and assessment as set out in this BPG is to determine the nature and severity of this potential impact. Mines that have an insignificant impact will be able to demonstrate this too, through application of the methodology set out in this BPG.

It needs to be recognized that this BPG will be used by different types of users with different levels of expertise and different needs. These different users are listed in

order of priority from the perspective of this BPG, as follows:

DWAF Official: this is the primary user and the primary purpose of this BPG with regard to the DWAF user is to give them an understanding of the process and procedures involved in an impact prediction exercise and to indicate where DWAF must be involved to give guidance and to agree on key criteria. However, it is assumed that the average DWAF user will not be a geochemist and it is therefore not intended that the DWAF user play the role of a specialist.

Project Reviewer: this is the secondary user as the reviewer is the person who will provide an independent specialist review and opinion that can be used by DWAF as input to the decision-making process. The role of the reviewer is to ensure that the impact prediction has been undertaken in a scientifically valid manner, that all assumptions that were made are reasonable and defensible and that all pertinent information (negative or positive) that could influence DWAF's decision is contained in the project documentation.

Mine: this is the tertiary user and the BPG is intended to give the mine clear guidance on the methodology that will be acceptable to DWAF and that can be used by the mine to develop the scope of work for the specialist's appointment and to ensure that the specialist undertakes his/her work in the correct manner.

Prediction Specialist: there are a range of specialist disciplines involved in an integrated impact prediction exercise and it is assumed that an appropriately qualified specialist, who knows that he/she will be continuously reviewed by an independent reviewer, can be relied upon to ensure that they are capable of undertaking the work. As such, it is also assumed that the specialist is familiar with their area of expertise and do not require detailed and specific instruction from a BPG. In fact, a suitably qualified specialist will be able to modify his/her approach to take account of site-specific conditions while still ensuring that DWAF's primary requirements are met.

This BPG is structured as follows:

- Chapter 2 covers the general principles for impact prediction.

- Chapter 3 presents and summarizes the basic risk assessment methodology (also accepted in the DWAF Water Use Authorisation Application Process) that underpins all impact predictions.
- Chapter 4 presents the Impact Prediction Process that should be followed and indicates the key points where DWAF officials need to be consulted and where decisions need to be made.
- Chapter 5 presents specific guidance on the types of questions that DWAF requires answers to from an impact prediction exercise at different stages of the mine life cycle and also provides guidance on the types of tools that should be considered when answering the different questions in order for decisions to be made.
- Chapter 6 describes the capabilities and limitations of different tools and techniques (including design and implementation of sampling) available for impact prediction of source, pathway and receptor.
- Chapter 7 presents a discussion on the issues of uncertainty and how these should be addressed and quantified within the impact prediction methodology and the presentation of results.
- Chapter 8 discusses considerations relating to independent review of the technically complex prediction exercise and how this review process should be implemented.
- Chapter 9 presents recommendations on the contents of an impact prediction report and a simple checklist that can be used by DWAF officials in reviewing and impact assessment.
- Appendix A will provide a discussion of the physical, chemical and biological processes involved in acidic, neutral and/or saline drainage generation and neutralization as well as groundwater flow and contaminant migration.
- Appendix B presents a detailed discussion on the various analytical, assessment and predictive tools commonly employed in making impact predictions and assess their capabilities and limitations.
- Appendix C presents a detailed discussion on the issues that need to be considered when developing a conceptual model
- Appendix D presents a case study of a conceptual model report for a waste disposal facility
- Appendix E presents a case study of a conceptual model report for tailings disposal facilities, waste rock dumps and reclaimed waste rock dump footprints.

2

GENERAL PRINCIPLES OF IMPACT PREDICTION

The basic principles that underpin the impact prediction methodology and which form the basis of the guidance provided in this BPG are:

- The level of confidence required from an impact prediction will vary depending on the nature of the mining scenario being evaluated, the life cycle phase of the mine, the consequence of an error in the prediction and the reversibility of management actions taken in response to the prediction.
- Suitably qualified persons must be utilized to undertake the impact prediction. They must have access to and knowledge of a full toolbox of impact prediction tools, must know their capabilities and limitations and must be able to fully motivate the methodology used.
- The traditional and established risk assessment methodology (source term, pathway and receptor) must be used in order to develop an appropriate conceptual model of the scenario to be evaluated – ensure that this step is discussed with and agreed to with all persons who will be reviewing the results of the assessment.
- Ensure that the correct and appropriate tools are used to answer the particular questions formulated for the particular mining scenario to be evaluated
- Use a number of complementary tools and compare outcomes to give convergent results.
- Understand the need for proper data collection and that confidence in prediction is dependant on quality of data and use of correct tools.
- Understand and define the uncertainty inherent in the impact prediction, based on composite uncertainties of the data collection process, the assumptions made, and the limitations of the tools used.
- Make provision for thorough independent review wherever the consequence of decisions based on the prediction is high and/or irreversible.
- Ensure that all impact predictions are followed by a monitoring programme designed to provide data for a future calibration / validation of the predictions originally made.

3

RISK-BASED
APPROACH
TO IMPACT
PREDICTION

The risk-based approach to impact prediction is favoured by DWAF and is consistent with policies and approaches that are subscribed to by DWAF in the review and approval of water use licence applications. It is recognized by DWAF that any prediction of future impacts has inherent uncertainty which means that there is always a risk that the prediction proves to be incorrect due to the occurrence of some unforeseen future event.

However, in order to accept the risk-based approach and the consequences that go with it, this BPG sets out very specific requirements that must be complied with in order to ensure that the risk is a manageable one. For this reason, the BPG also defines a specific methodology and requires the concurrent involvement of an independent reviewer whenever the impact assessment is used as part of a mine closure application. The defined methodology also requires specific consideration and definition of uncertainties within the assessment process.

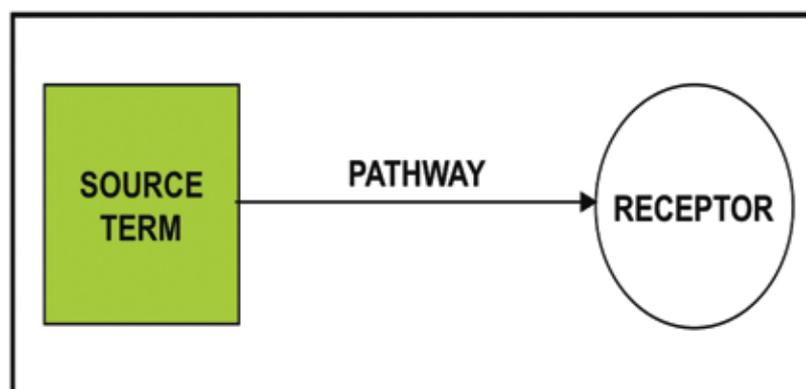
In cases where mining has occurred on an intensive regional scale with hydraulic interconnections between adjacent mines, it is also necessary to specifically consider these regional interactions, especially when it comes to making impact predictions in support of mine closure applications. In this instance, readers are referred to Pulles et al, 2005 where a discussion and methodology for the consideration of regional impacts is presented.

The general principle inherent in the mine closure risk assessment methodology is that the mine must take responsibility for all risks that can be foreseen, by way of a post-closure financial provision, DWAF accepts the risks associated with unforeseen events, provided that the impact prediction process complies with the requirements set out in this BPG and in **BPG G5: Water Management Aspects for Mine Closure**.

3.1 GENERAL RISK-BASED APPROACH CONCEPTS

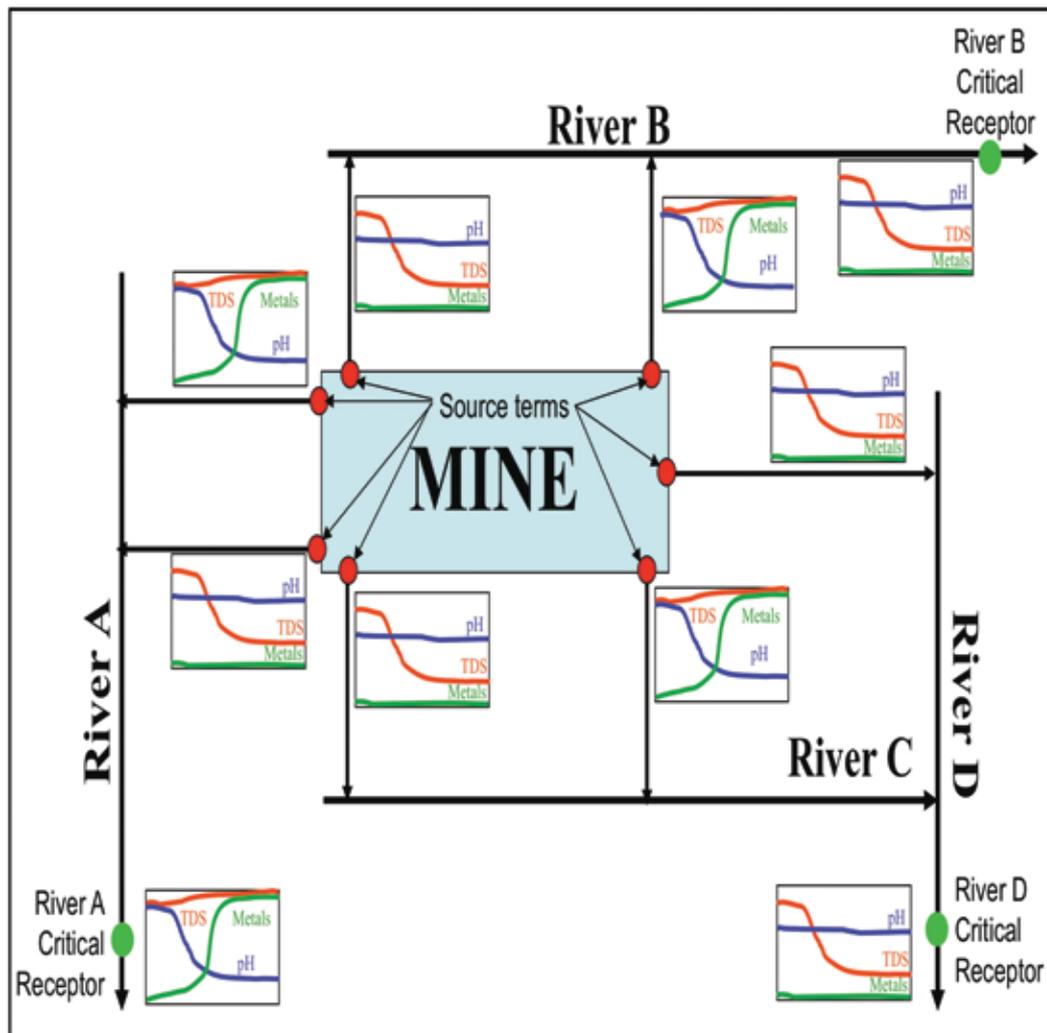
Risk assessment entails the understanding of the generation of a hazard, the probability that the hazard will occur and the consequences if it should, i.e. understanding of the complete cause and effect cycle. The most basic risk assessment methodology is based on defining and understanding the three basic components of the risk, i.e. the source of the risk (source term), the pathway along which the risk propagates, and finally the target that experiences the risk (receptor). The risk assessment approach is aimed at describing and defining the relationship between cause and effect. The basic risk-based approach is shown schematically in Figure 3.1 below.

Figure 3.1: General risk assessment components



A complete risk assessment will assess all three components in all their various forms. While Figure 3.1 shows a single source term with a single pathway and single receptor, in reality, multiple source terms, pathways and receptors exist and a full mine site risk assessment may require a fully integrated assessment where the various source terms, pathways and receptors are considered together. However, such a fully integrated risk assessment would be technically very complex and a more rational approach would be to undertake an initial screening level assessment and to then determine the critical receptor for a particular source term which would be the most severely impacted upon, to agree on the defined critical receptor with DWAF and to focus the assessment on determining the risk to that critical receptor.

Figure 3.2: Risk Assessment for Multiple Source Terms & Multiple Receptors



Furthermore, many assessments are only interested in evaluating the risk from a single source term, for example, the assessment of cover alternatives for a coal discard dump would only be interested in the discard dump as the source term and then evaluating the impact on the selected critical receptor for the various pathways that exist. In another typical example (see Figure 3.2 above), a complete underground mine with various decant points may serve as the source term, the receiving watercourse could be the pathway and a downstream irrigation user may be the critical receptor. In such a pre-defined scenario, the assessment is greatly simplified, although care must be taken, through an appropriate screening level assessment, to ensure that the irrigation farmer is, in fact, the critical receptor rather than, for example, an endangered endemic fish species in the receiving watercourse.

3.2 DEFINING THE SOURCE TERM

In the context of predictions of impact on the water resource at mining sites, the source term could include any of the following:

- Underground mine void
- Opencast pit
- Waste rock dump
- Coal discard dump
- Tailings disposal facility
- Ore and product stockpiles
- Neighbouring mines
- Any other potential source of impact

It is important to note that the various source terms described above are all very complex facilities when it comes to confidently predicting their behaviour into the future. The future behaviour of the source terms is determined by two primary driving forces:

- the geochemistry of the material within the reaction pathway, and
- the hydraulic characteristics of the source term which liberate and mobilize the chemical reaction products.

The approach to defining the behaviour of the source term will always start with the definition of the key questions that need to be answered for the source term (see Chapter 5), followed by the development of a detailed conceptual model that describes the key driving

forces that act upon the source term. The conceptual model must also ensure that the necessary information is defined to support the tools that need to be used to answer the stipulated questions.

The detailed assessment that will follow from the conceptual model will need to make use of a combination of the following tools:

- Hydraulic infiltration models
- Geohydrological models
- Water balance models
- Oxygen diffusion and convection models
- Geochemical sampling and analytical programme
- Geochemical models (oxidation rate, speciation and mass balance)

The output from the source term characterization will typically be presented in the form of graphs (with datasets) that show how seepage volumes and quality (for the various contaminants of concern) vary into the future. It needs to be recognized that source terms are mining features that are dynamic in nature and that exhibit a variable quality over time, due to changes in hydrology and to changes in the chemistry as sulphide minerals or neutralizing minerals become depleted or vary in reactivity, or as secondary minerals precipitate or redissolve as conditions change. An impact assessment that defines the source term as a static constant feature over time is unlikely to be realistic and would be inappropriate for anything other than the most basic screening level assessment.

3.3 DEFINING THE PATHWAY

In the mining context and with respect to potential impacts on the water resource, the pathway through which contaminants could move would most typically be one or more of the following:

- Movement through the vadose (unsaturated) zone
- Movement through an aquifer
- Movement through surface runoff in storm water or a watercourse
- Movement through mining voids (underground or opencast)
- Airborne migration of sulphide minerals or other contaminants as dust

As with the source term, the first step in defining the pathway would be to take cognizance of the questions that need to be answered (Chapter 5) and to then construct

a suitable detailed conceptual model that defines the various pathways of interest and the variables and factors that need to be considered when assessing these pathways. Within the context of defining the pathways it is important to note that the pathways may have the following features:

- a hydraulic conduit (pathway) for the mobilization and movement of the contaminants of concern from the source term to the receptor
- attenuation of contaminants, release of new contaminants and alteration of the chemistry of the discharge from the source term through a variety of chemical reactions
- habitat for receptors

The detailed pathway assessment that will follow from the conceptual model may need to make use of a combination of the following tools (see Chapter 6):

- Geohydrological models (possibly with consideration of attenuation mechanisms)
- Hydrology / runoff models
- Water quality models for surface water systems

The output from the pathway characterization will also typically be presented in the form of graphs (with datasets) that show how water quality (for the various contaminants of concern) varies into the future at the critical receptor. Given the fact that hydrology changes on a seasonal and annual basis, it follows that these changes will also affect the water quality at the receptor. It will normally be important to understand which hydrological conditions will result in the worst case scenario for the receptor of interest and to understand the statistical frequency with which such scenarios could occur.

3.4 DEFINING THE RECEPTOR

As the final component of the risk assessment, the receptors in the context of the water resource would be users of the water resource itself and typical examples could be the following:

- Groundwater user abstracting contaminated groundwater through a borehole for domestic, livestock watering or irrigation use
- Aquatic fauna and flora in a receiving watercourse
- Any water user abstracting water from an impacted watercourse

While the effect of various contaminants of concern on the various receptors would typically be evaluated

through the use of various dose-response models, the common approach in environmental risk assessments would be to rather use published or regulatory water quality criteria for the receptor (user) of interest. DWAF has a series of water quality guidelines for the various recognized water users that do incorporate consideration of dose response information and unless a very unique critical receptor is identified for a mine risk assessment, it would be appropriate to make use of the published DWAF water quality guidelines and any specific resource quality objectives that derive from a reserve determination or that derive from consultation with affected water users/receptors. These guidelines tend to be conservative and relaxed guidelines would typically require a more detailed risk assessment, using typical dose-response modelling to show that the risk to the critical receptor remains acceptable. The agreement on the guidelines and quality objectives that should apply at the critical receptor must involve DWAF officials in addition to other water users where appropriate.

As it is generally impractical and unnecessary to consider the full range of potential receptors that may be impacted upon by any particular source term, it is appropriate to define a critical receptor – which is usually that water user which is the closest to the source term or which is the most sensitive to contaminants produced by the source term. In determining the critical receptor it is also necessary to consider potential future receptors. The critical receptor should be clearly defined in the conceptual model and should then be agreed upon with the affected parties and DWAF before the risk assessment and impact prediction is undertaken. Such an interaction with DWAF could be considered as complying with Stage 1: Screening and application as defined in DWAF's Internal Guideline for the Water Use Authorisation Application Process.

3.5 IMPORTANCE OF THE CONCEPTUAL MODEL

As discussed above, the first and most critical component of a risk and impact assessment is the development of a suitable detailed conceptual model for the source terms, pathways and receptors. The conceptual model is arguably the most important step in the whole impact prediction exercise as it defines the questions that need to be asked, the design of the sampling programme, the tools and techniques to be applied in the prediction and the various assumptions and data values that will be used in the project.

It is considered absolutely critical that an initial conceptual model be fully developed, described and documented in a conceptual model report that must be signed off on by the project reviewer and DWAF before any further work, such as sampling, analyses, data review, modelling, etc. is allowed to proceed. If the conceptual model is flawed then the whole prediction exercise will also be flawed. The initial conceptual model must be based on site visits, review of available data and must physically describe the mining feature (source term) in terms of those aspects that would have a bearing on the impact prediction. It must also describe the pathways along which the impact will propagate and then describe the procedure that was applied to determine who/what the critical receptor is. Examples of typical conceptual model reports are given in Appendices D and E.

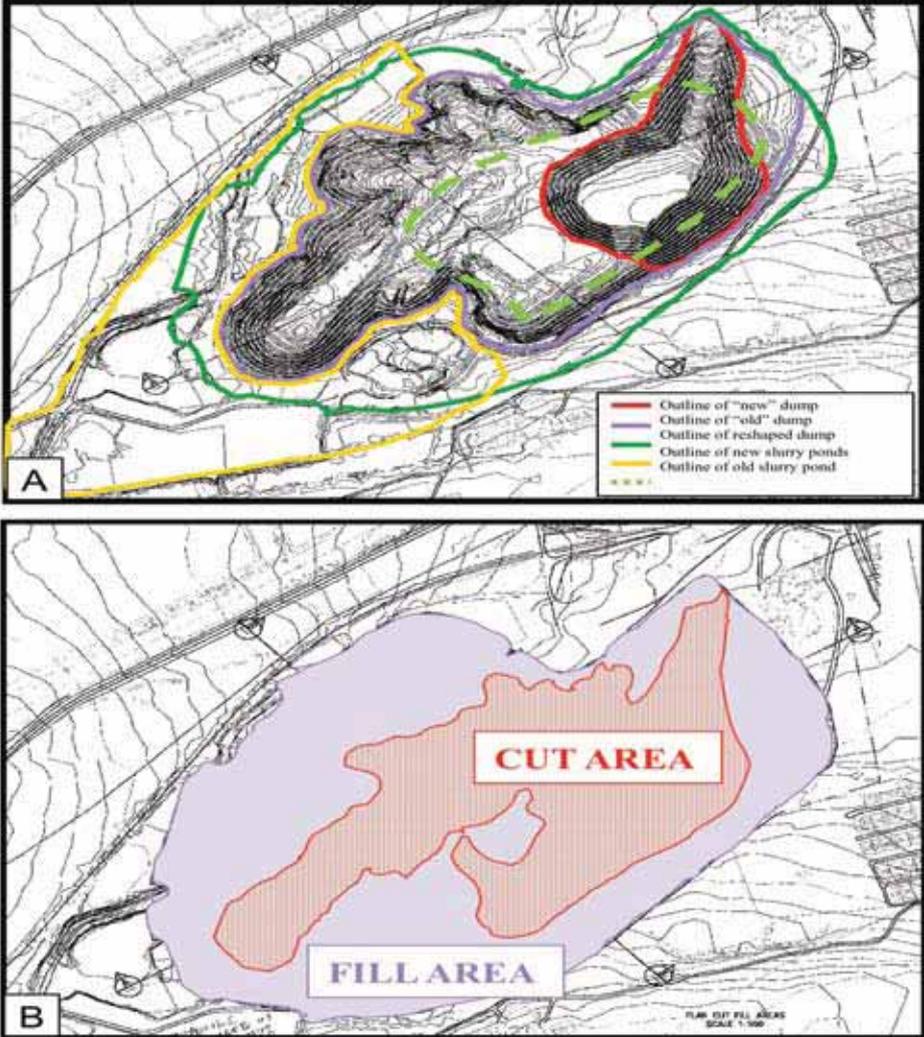
The complexity of the initial conceptual model is also determined by the type of study being undertaken, e.g. a simplified conceptual model for a screening level study and a detailed conceptual model for a mine closure application. The initial conceptual model is used to define and develop a sampling and monitoring programme and once the data has been obtained, the initial conceptual model and report will be updated and issued as a final conceptual model and report – again for signoff by the project reviewer and DWAF.

An example of a physical conceptual model for a partially burnt coal discard and slurry disposal facility (source term only) is shown in Figure 3.3 above. Figure 3.3A shows a plan view of the dump with outlines of different areas of the facility, distinguished in terms of their pollution potential. It also shows the outline of the final reshaped dump. Figure 3.3B shows the plan view of the cut and fill that will need to happen to get the dump to its final shape. Figure 3.3C shows the different nodes of the facility, distinguished in terms of their pollution potential, for the existing dump, while Figure 3.3D shows the same after the dump has been reshaped. Each of the nodes shown in these figures is then fully described in the conceptual model report in terms of physical, hydrological and geochemical characteristics and the nodes are then linked together in terms of a water flow path to serve as the basis for the mathematical model. The sampling programme will then also be structured to ensure that the different materials in the different nodes are sampled in a representative manner.

A further example of a simplified conceptual model for an integrated coarse and fine waste disposal facility is shown in Figure 3.4. Additional detailed information on

issues that need to be considered in the development of a conceptual model are shown in Appendix C.

Figure 3.3: Example of physical conceptual model



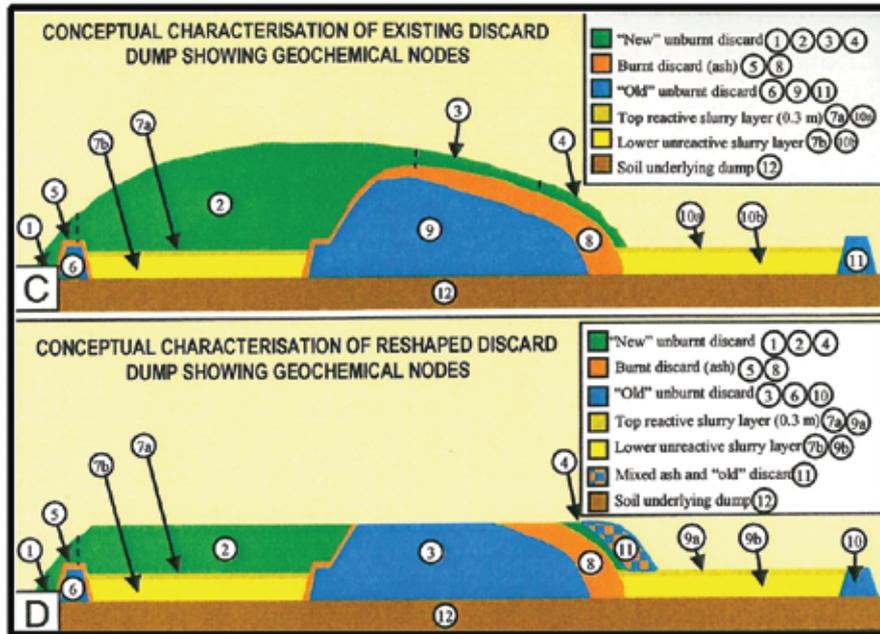
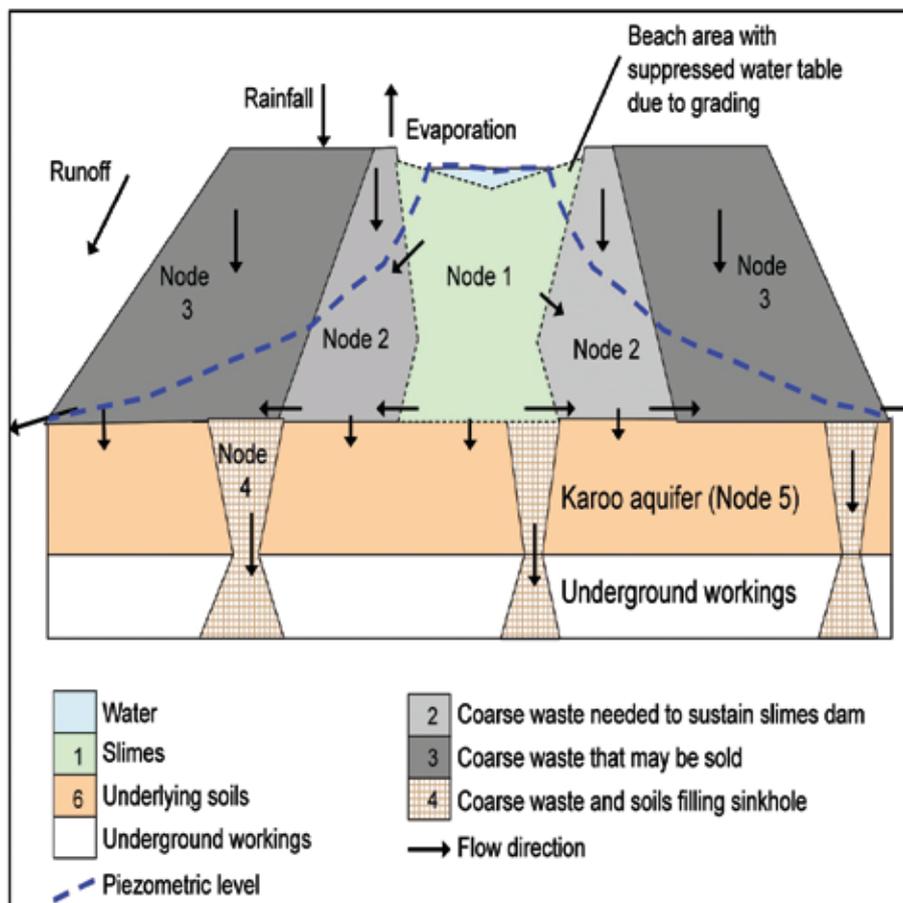


Figure 3.4: Simplified Conceptual Model



4

IMPACT
PREDICTION
METHODOLOGY
& PROCESS

The impact prediction process includes the following primary roleplayers:

- The mine and its appointed specialists
- The appointed independent reviewer
- DWAF personnel

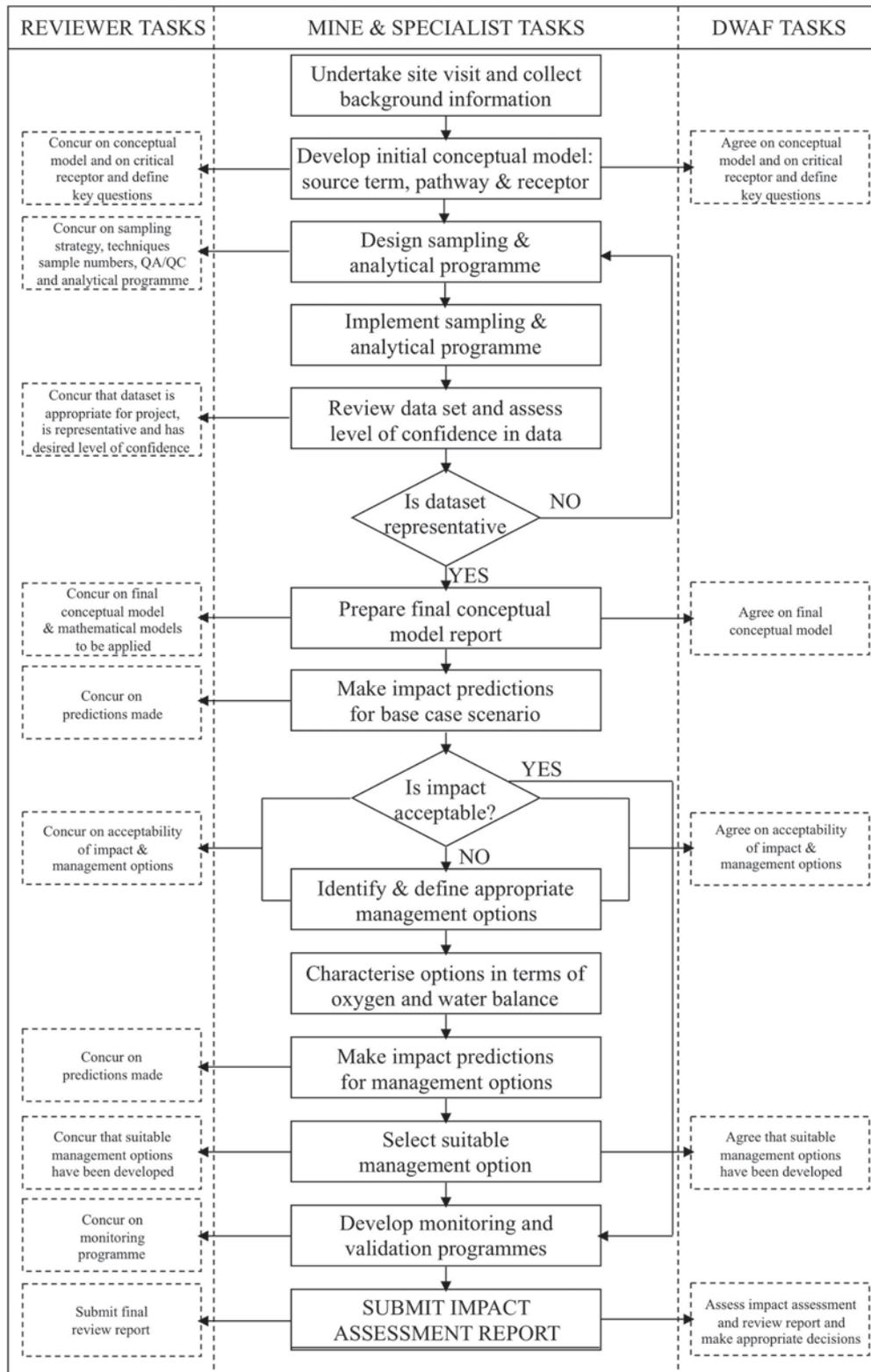
Each of these parties have different roles to play and different tasks to fulfill in the impact prediction exercise as shown and summarized in Figure 4.1 below. This Figure shows the typical generic impact prediction methodology that a specialist would be following in undertaking a detailed quantified impact prediction. While the precise methodology that a specialist may choose to use may vary slightly depending on site-specific scenarios and considerations, the basic methodology is valid for all impact predictions.

The key principle to be obtained from Figure 4.1 is that there are very specific points along the impact prediction process where the mine and its specialist **must** engage with and reach consensus with the independent reviewer **before** proceeding to the next step in the project. Failure to obtain this consensus could be viewed as a fatal flaw in the impact prediction exercise. The role of the reviewer is also discussed in more detail in Chapter 8 of this BPG.

Additionally, there are a number of key decision points where the mine, its specialist and the appointed independent reviewer must meet with and engage with representatives from DWAF in order to obtain agreement from DWAF on key issues. In this context, the term 'agreement from DWAF' will take the form of **written minutes of the meeting, formally approved and signed off** by the mine, its specialist, the independent reviewer and DWAF at the subsequent meeting. The very last step, i.e. review and approval of the impact assessment report by DWAF will be by way of written approval by a suitably authorized person at DWAF on a DWAF letterhead.

After the impact assessment has been undertaken, there is a need to implement the monitoring and validation programmes and after a suitable period (typically 3-5 years), review the impact prediction that was undertaken, particularly the numeric models and data sets that were used and then validate the accuracy of the predictions. The key point here is that the validation and calibration process must critically review the earlier modeling and issue a statement of assurance in the accuracy of the earlier predictions. This statement of assurance must also specifically address the reliability of the long-term predictions in terms of duration and magnitude of the predicted impact.

Figure 4.1: Impact Prediction Methodology & DWAF/Reviewer Roles & Tasks



5

KEY IMPACT
PREDICTION
QUESTIONS

The first step in undertaking an impact prediction exercise, is to define the questions that need to be answered with the prediction, in order that the correct assessment and modelling tools can be selected. This document provides guidance on which type of questions need to be answered for different assessment scenarios. One of the key distinguishing features when it comes to defining the questions to be asked and answered is whether or not the impact prediction is being undertaken to support a mine closure application. In the event of mine closure the issue is essentially about transferring risk and potential liability from the mine owner to the State and taxpayer and there is therefore a requirement for high confidence in a quantitative prediction.

The nature of questions to be answered also depends on the life cycle phase of the mine as it stands to reason that fully quantitative and detailed assessments of post-closure impacts cannot be undertaken during the mine feasibility planning stage as there is still too much uncertainty regarding mine layout, waste characteristics, nature of the water resource, etc.

5.1 PROSPECTING, MINE FEASIBILITY AND MINE PLANNING PHASES

Key questions that should be asked and answered at the prospecting, feasibility and mine planning phase of a mine project would include the following:

- 1) Will any waste material be generated that has a potential to generate acid, neutral or saline mine drainage? (*Tools to be used include sampling of exploration drill cores, bulk samples and waste streams from pilot studies and application of static and kinetic analytical procedures to these samples, as well as comparison to actual data for other mine sites exploiting the same ore body. May consider simplified mathematical modelling*).
- 2) Is there a potential to separate and manage waste streams in accordance with their acid, neutral or saline drainage potential? (*This component should be evaluated in conjunction with mine planning personnel*).
- 3) Are there any potential positive or negative consequences of storing and/or disposing of these waste materials in any particular fashion, on their own or in any combination? (*Use analytical data collected to answer Question 1 above and use in appropriate mathematical models to predict long-term consequence of different waste disposal options and use results as input to decision making on waste disposal option to be implemented*).
- 4) How would proposed alternative mining techniques and layouts (backfill material for opencast pits) affect the potential impact on the identified receptor water resource (surface and groundwater balance and quality)? (*Use detailed hydrological and geohydrological modelling to evaluate effects of alternatives on water balance in the mine and impacts on the water resource. Apply simplified (equilibrium and speciation) mathematical modelling to determine probable mine drainage quality in order to define potential need for water treatment*).

5.2 MINE OPERATING PHASE

Key questions that should be asked and answered at the operational phase of a mine project would include the following:

- 1) What is the long-term impact of all waste residue deposits (fine and coarse waste) on the water resource (surface and groundwater) in terms of volumes and quality of drainage

over the life of mine and post-closure? *(Undertake statistically representative sampling of all waste residue deposits on the mine and subject samples to static and kinetic analytical programme in order to determine variability within and between waste deposits. Use simplified kinetic geochemical models to predict long-term water quality profiles for contaminants of concern).*

- 2) What final rehabilitation should be undertaken for the different waste residue deposits in order to meet long-term risk management objectives for the water resource? *(Using data collected for Question 5 above, and using the simplified kinetic geochemical model, evaluate the effects of the various rehabilitation and closure options in terms of impact on the source term. Undertake detailed physical modelling (water and oxygen balance) of the various alternatives and use the physical model outputs in the simplified kinetic geochemical models. The focus should be on comparative results rather than on absolute values).*
- 3) Will the mine void (pit or underground mine) decant after mine closure? If yes, where, when, how much and at what quality over time? *(The assessment of whether or not the mine will decant into either the aquifers or into the surface water resource will be determined on the basis of **detailed** mine water balance modelling incorporating hydrological and geohydrological models. If the mine is predicted to decant, then decant points, time of decant and volume of decant will need to be determined using the same models. Once the flow balance modelling has been undertaken, a kinetic geochemical modelling component will need to be added to determine water quality profiles over time for all the decant points and for all the contaminants of concern. Using this as a base case situation, a range of management options to reduce the volume and/or improve the quality of the predicted decant must be defined and evaluated using the detailed water balance and geochemical models).*

A key component of impact prediction in the operating phase of the mine is that it presents the opportunity to collect data to use to calibrate and validate the predictive models. At least 5 years before planned mine closure, all impact predictions should proceed to the level of detail required for the mine closure phase. As the above questions and the studies that will need to be undertaken

to answer the questions all relate to the identification and assessment of management measures that will persist after mine closure, the detailed independent review of the assessments is strongly recommended.

5.3 MINE CLOSURE PHASE

Key questions that should be asked and answered at the closure phase of a mine project would include the following:

- 1) What are the drainage volumes and quality for all contaminants of concern for all source terms that pose a potential risk of impacting on the water resource – such profiles to show predictions at least 100 years into the future, or longer if longer periods are required to quantify the impact, as recommended by the specialist and agreed to by the independent reviewer and DWAF. *(The assessments described for Questions 1-3 for the operational phase of the mine will need to be extended to ensure that boundaries of confidence can be defined for the predictions and that all models are fully validated and calibrated and that full independent review has been undertaken).*
- 2) What will the long-term impact be at the critical receptor for the contaminants of concern? *(The various source term assessments will need to be linked into an integrated model that incorporates regional groundwater and surface water hydrology and quality and that presents detailed time-based water quality profiles at the critical receptor for at least 100 years or until the water quality at this point has reached stable conditions).*
- 3) What additional water management (e.g. covers, infiltration reduction measures, etc.) or treatment measures need to be instituted to reduce the contaminant loads from the various source terms or to intercept the pathways in order to ensure that the critical receptor is not adversely impacted? *(Using the models developed to answer closure phase questions 1 and 2 above, apply and evaluate the effects of various mitigation measures in terms of impact at the critical receptor).*

While it is critical that the impact prediction assessments undertaken for the purpose of mine closure applications undergo a full independent specialist review, it is also important that such closure impact predictions be undertaken using computer code and mathematical

models that are available in the public domain, in order that any third party can review and independently reproduce the modelling results. Proprietary code should be avoided for closure impact predictions.

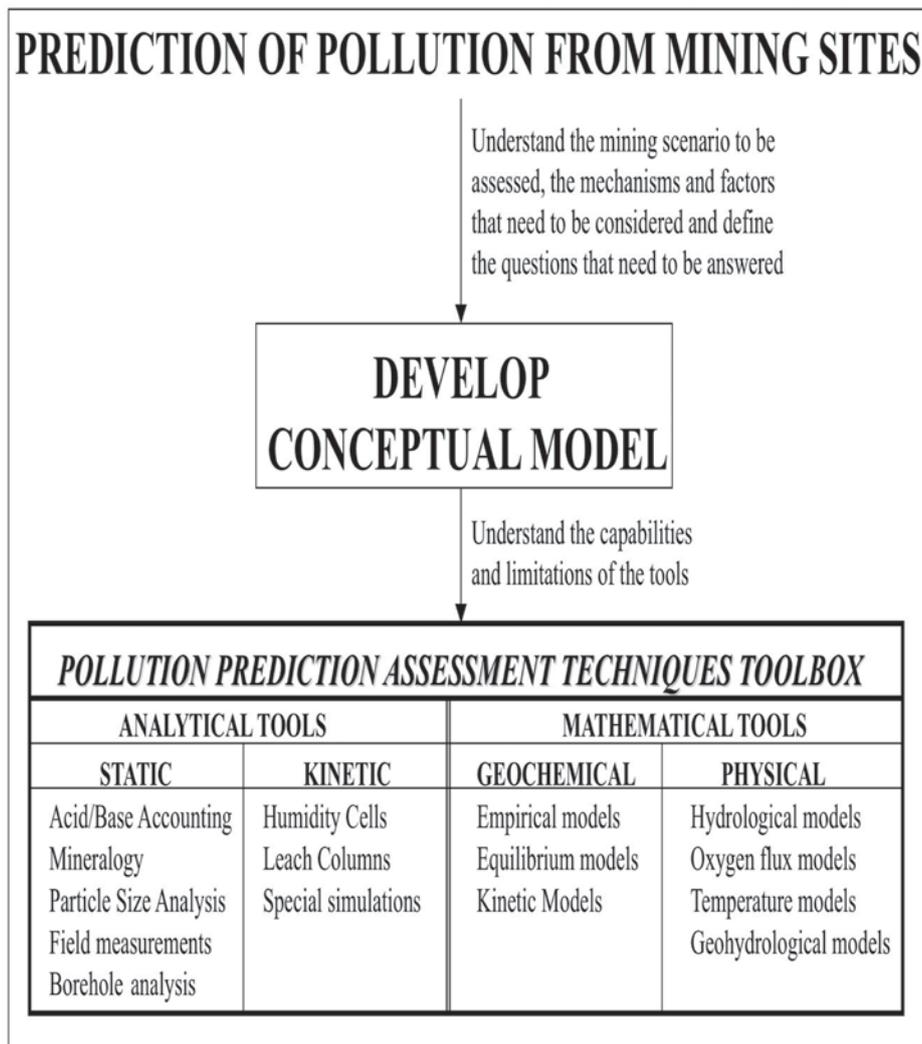
5.4 WHAT ARE APPROPRIATE MODELS AND COMPUTER CODE TO USE?

While the above questions in Sections 5.1 to 5.3 do indicate what class of tools should be used to answer particular types of questions (for example, where kinetic geochemical models should be used as opposed to equilibrium models), the basic guiding principle that should be applied is that the independent reviewer should be party to the decision-making on what type of analytical tools, conceptual models, mathematical models and computer codes should be used for any given assessment, rather than attempting to be prescriptive in this BPG in a manner that caters for all the site-specific eventualities found on mine sites.

6
**IMPACT
PREDICTION TOOLS
& PROCEDURES**

In order to be able to make a prediction of future pollution from mining sites, it is necessary to go through the process, as shown in Figure 6.1 below.

Figure 6.1: Pollution Prediction Process and Assessment Techniques



A key component of the prediction process is use of the “toolbox” of assessment tools and techniques. It is important to emphasize the following points at the outset:

- 1) The assessment process is complex and requires use of a range of sophisticated tools. Competent practitioners must be capable of demonstrating their ability to use the full range of tools and not only one or two of the tools in order to ensure that the right tools and techniques are being applied to the problem at hand.
- 2) The modelling tools should only be applied by suitably qualified persons and the analytical tools should only be applied by reputable laboratories. It is unlikely that any one person can be suitably qualified to apply the full range of models that include geohydrology, hydrology, oxygen and heat transport, equilibrium and kinetic geochemical models and a team approach employing the appropriate combination of skills is most often required.
- 3) The assessment process must include a suitable quality assurance and quality control (QA/QC) programme. For the analytical assessment techniques this QA/QC programme would typically include a provision for duplicate samples, use of alternative analytical techniques and the employment of statistical interpretation techniques to ensure that datasets are statistically representative. For the modelling techniques, the QA/QC programme would typically start with ensuring that the appropriate specialists and models are used and could also include external independent review, particularly for complex quantitative assessments.
- 4) Any predictions that are made need to be verified with an appropriate monitoring programme and the collected data must be used to calibrate the assessment techniques and models that were used.

The assessment tools and techniques that are needed to make a prediction of pollution from mining sites can be categorised in different ways. For the purpose of this BPG, the categories as shown in Figure 6.1 above are proposed and the primary distinction is made as follows:

- Analytical tools (static and kinetic)
- Mathematical tools (geochemical and physical)

An important distinction between these two types of tools is as follows:

- Analytical tools generally have higher precision and repeatability but have limited predictive ability (especially the static techniques), they also provide data that serves as input to the predictive mathematical models;
- Mathematical models do have a predictive ability (when the right models are used for the aspect being assessed) but have lower and often uncertain precision, especially if poor datasets are used and proper model calibration has not been undertaken.

The impact prediction process will need to make use of a suite of different tools – analytical methods and models - and the appropriate tools for each situation will need to be clearly defined in the conceptual model report. Where an independent reviewer is involved in the project, he/she will need to agree on the procedures and tools used in the impact assessment.

A more detailed discussion on the capabilities and limitations of the various tools is presented in Appendix B to this BPG.

7

PRINCIPLES
OF UNCERTAINTY

It must be clearly understood that uncertainty is inherent in any prediction exercise and does not represent a fatal flaw with the methodology. Any future prediction of water impacts is based on assumptions about future rainfall, data values that are approximations of reality and tools that attempt to describe natural processes as mathematical formulae. While it is accepted that uncertainty is inherent in any prediction, the important issue is that the specialist undertaking the prediction must be able to describe and define the uncertainty in the prediction, in order that margins of safety can be built into management options and/or financial provisions.

7.1 TYPES OF ERROR AND UNCERTAINTY

Any impact prediction exercise will encounter numerous sources of uncertainty, such as the following:

- **Sampling uncertainties** - resulting from non-representivity of samples, such as incorrect spatial distribution of samples, insufficient number of samples to cater for natural heterogeneity, errors in sampling technique (e.g. bias towards fines or coarse material).
- **Sample storage and preparation uncertainties** – due to changes in nature of the sample due to incorrect storage techniques and bias associated with poor sample preparation within the laboratory where a subset of the field sample is taken for actual analysis.
- **Analytical uncertainties** – due to quality assurance / quality control errors within the laboratory, operator error within the laboratory and errors and detection limits associated with the analytical techniques themselves.
- **Assumption and estimate uncertainties** – due to errors in the assumptions that are made and incorporated within the conceptual model, and the estimates that are made where precise and accurate data is not available (for example, where different types of waste are mixed within a waste residue deposit, it may be necessary to make assumptions on the effectiveness of the mixing, or the uniformity of layers, or where covers are modeled, certain assumptions are made with regard to the homogeneity of the cover).
- **Hydrological uncertainties** – all impact predictions need to utilize hydrological records and make assumptions on future rainfall events. Standard practice is to use historical hydrological records and to assume that future rainfall patterns will be similar and this assumption has obvious inherent errors, further compounded by the imprecise future consequences of global warming.
- **Extrapolation uncertainties** – A number of kinetic tests such as humidity cells and leach columns generate datasets which can be plotted as contaminant profiles over a time period. Some assessments are based on extrapolation of such data into the future which gives rise to errors as the test simulation conditions are not equivalent to real-life field conditions.
- **Mathematical modelling uncertainties** – future predictions rely heavily on mathematical models – all of which are simplified versions of reality. Whereas some models have very precise chemistry and mineralogy routines and simplified mass transport and flow routines, other models do the opposite. The key point is that no model can be viewed as a precise simulation of reality and all model results have inherent error within them.
- **Model coupling uncertainties** – a fully integrated impact prediction exercise will utilize a range of different models with the outputs from one serving as inputs to the next model. For example, a hydrological runoff model may feed into a groundwater flow model to determine inflow into an underground mine. Once the water is in the mine, the calculated volume of water is subjected to a geochemical model to determine its water quality changes and to predict a volume and quality of mine decant. The decant is then again used as input to a river flow and quality model to predict water quality changes at a downstream critical receptor.

Each of the models in this example uses and outputs data in very different formats and integrating these models is a complex task that generates a number of uncertainties.

7.2 SENSITIVITY ANALYSES AND PROBABILISTIC MODELLING

Not all the errors have the same consequence on the impact prediction and the most significant factors can be determined by undertaking sensitivity analyses where the predictions are rerun where all parameters remain constant except for one which is then varied around the average value used for the predictions. Repeating this exercise for a number of key variables will clearly

indicate which variables have the biggest effect on the prediction result. Undertaking a sensitivity analysis should be standard practice in any complex prediction exercise where uncertainty on key variables exists.

A second strategy that can and should be employed as a tool to better define the uncertainty in predictions is to undertake the mathematical modelling within a probabilistic or stochastic framework (also known as Monte Carlo modelling) where key variables are not represented in the model as a single average value, but where values are randomly selected from a known distribution of values for each parameter. The model is then rerun numerous times for each unique set of variables and the results are plotted as an average value with confidence limits showing upper and lower boundary values as shown in Figure 7.1 below.

Figure 7.1: Schematic Representation of Probabilistic Modelling

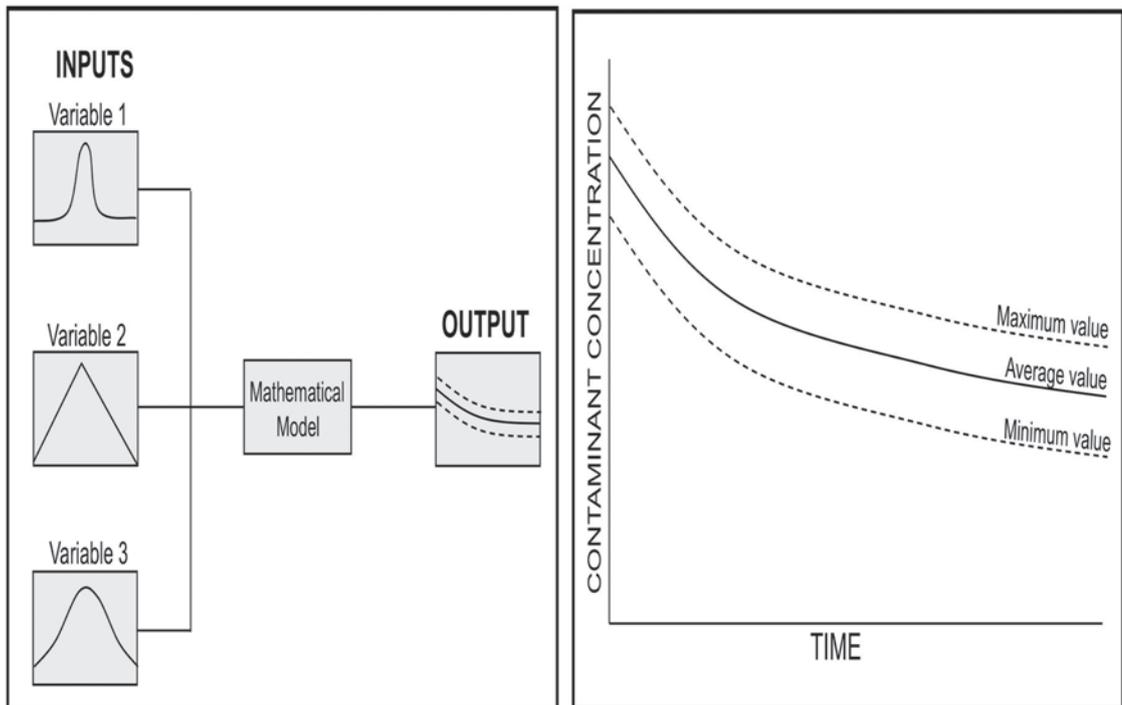
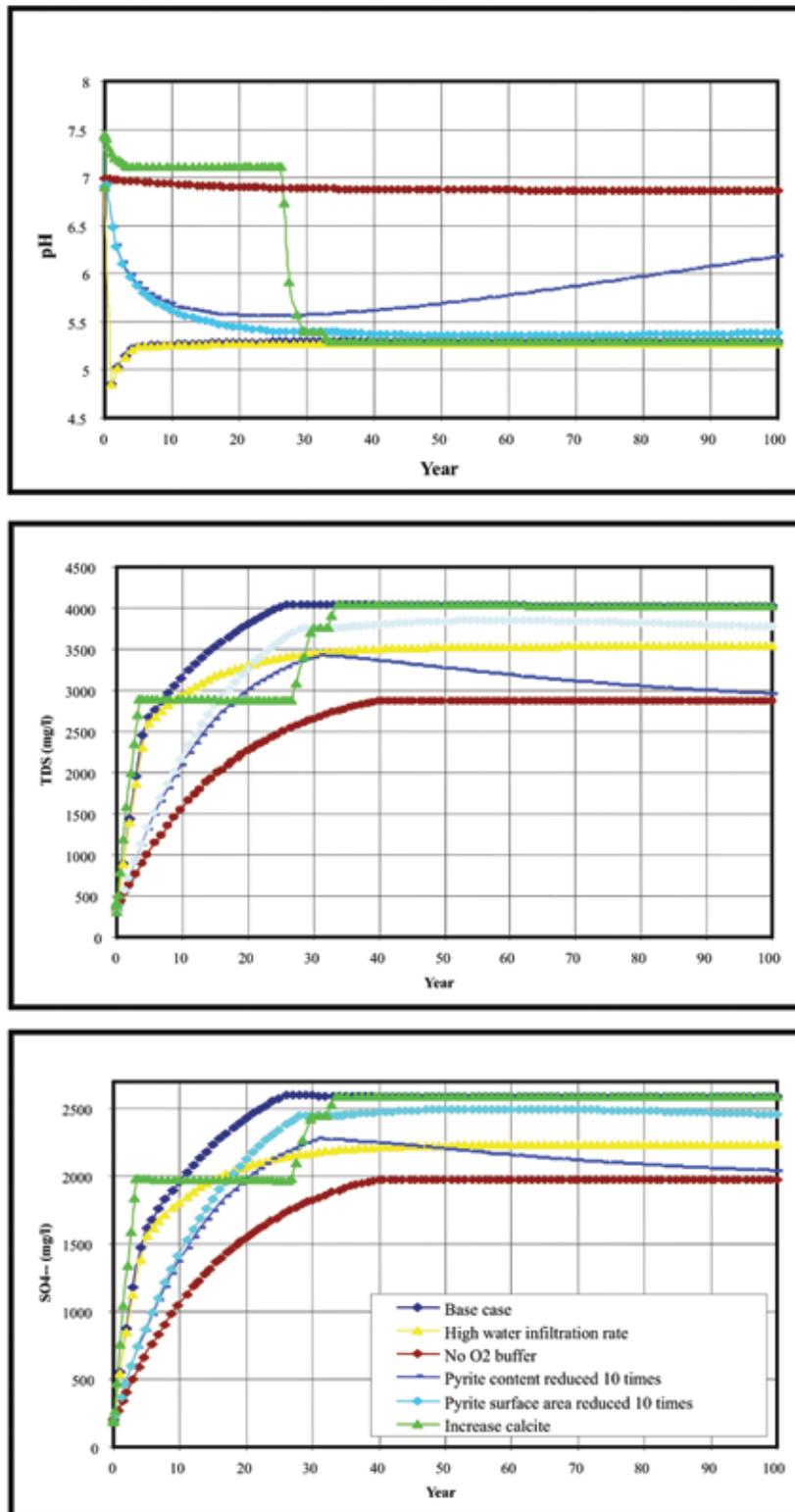


Figure 7.2: Example of sensitivity analyses for a tailings disposal facility



An example of the outcome of a limited sensitivity assessment for an impact assessment undertaking for a tailings disposal facility is shown in Figure 7.2 above.

7.3 DEFINING ACCEPTABLE CONFIDENCE LIMITS

The definition of acceptable confidence limits and an acceptable degree of uncertainty is a difficult yet important task. From the regulator's perspective, the acceptable confidence limit refers to the mine closure situations where the objective of mine closure is to transfer risk and liability to the State. The confidence limits indicate the probability that the end result will be outside the boundaries defined in the impact prediction exercise, e.g. for a 90% confidence limit, the chance of the real life situation at a future date being outside the predicted boundaries is 10%. Once this has been defined, the appropriate margins of safety for developed management actions and financial provisions can be identified.

8

INDEPENDENT
REVIEW

Impact predictions are technically highly complex and require the efforts of integrated teams of specialists – e.g. hydrologists, geohydrologists, and geochemists – and will be managed by Project Managers with many years of specialist experience. Successful review of an impact prediction exercise requires the expertise of persons who have gained similar expertise to the team undertaking the assessment. As these specialist skills (especially geochemists with relevant experience) are in very short supply, it is considered highly unlikely that the regulatory authorities will have the in-house expertise to properly review the impact prediction reports that are submitted for approval. In recognition of this reality, this BPG recommends an external independent review process using specialists that do have the appropriate expertise. It is therefore recommended that the following review process be applied in all cases which meet the following criteria:

- 1) Impact prediction is in support of a mine closure application
- 2) Impact prediction is in support of regulatory approval for a mining feature that will persist into the post-closure phase.

The proposed review process is intended to **provide surety both to the mine and DWAF** on the confidence that can be placed in the impact prediction and the consequential decision made based on the prediction (refer to Section 41 of the NWA) and must comply with the following:

- 1) The mining proponent should issue a notification of intent to undertake an impact prediction process to DWAF.
- 2) The mining proponent and DWAF should jointly agree on how the reviewer should be appointed. It is recommended that at least 3 potential independent reviewers should be jointly identified who can be approached to submit a proposal and that agreement should also be reached on what the scope of their appointment would be (also see Figure 4.1).
- 3) The reviewer will be required to declare his impartiality and lack of interest in the outcome of the assessment, in writing.
- 4) The reviewer should be appointed at the beginning of the prediction assessment and should, as a minimum, provide review and input into the following stages of the project: development of conceptual model, design of sampling and analytical programme, review of sampling data, agreement on mathematical models / computer code to be used, review and agreement on all key assumptions used in the modelling, review and agreement on scope of alternatives to be considered in the assessment, review and agreement on all project conclusions and recommendations; an assessment of the uncertainty and error in the impact prediction, review and agreement on post-modelling monitoring, model calibration and validation programme.
- 5) The reviewer will present his/her review findings to joint meetings attended by the mining proponent, the mining proponents appointed specialists and the regulators. All review comments and documents will be on record and in the public domain.
- 6) The reviewer will be allowed to undertake his/her review work without pressure or interference from any of the parties involved in or with an interest in the assessment and must have the specific involvement shown in Figure 4.1.
- 7) The reviewer must provide a written input to the final impact prediction report as shown and discussed in Chapter 9.

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CONTENTS OF AN IMPACT PREDICTION REPORT

While there are different ways in which an impact prediction report could be structured, the following contents are recommended:

Chapter 1: Description of scenario being assessed

Chapter 2: Objectives of assessment and questions to be answered

Chapter 3: Conceptual model

- Description of source term
- Description of pathway
- Description of critical receptor
- List of key assumptions and literature-based data inputs
- Assessment by project reviewer

Chapter 4: Sampling and analytical programme

- Description of sampling programme (number of samples, location of samples, sampling method, sampling preservation, sample storage, sample preparation for analyses)
- Description of analytical programme (analyses that were undertaken, required detection limits, QA/QC programme)
- Description of statistical interpretation of data and calculated confidence limits of data set
- Assessment by project reviewer
- Chapter 5: Data interpretation and review
- Review of all analytical data and presentation of data tables and graphs
- Preliminary impact assessment based on data interpretation
- Assessment by project reviewer

Chapter 6: Mathematical modelling

- Description of models to be used and motivation for use of selected models for base case and selected alternatives
- Description of model input files and databases used for base case and each alternatives
- Integration of models – challenges and solutions
- Presentation of modelling results and comparison with observed data
- Sensitivity analyses on key variables
- Probabilistic assessment
- Definition and description of assessment uncertainties
- Definition of detailed monitoring programme for collection of data for future model calibration / validation exercise
- Assessment by project reviewer

Chapter 7: Consolidated impact prediction

- Discussion of results and formulation of risk and impact on critical receptor and need for risk management / mitigation measures
- Assessment by project reviewer

Chapter 8: Conclusions and recommendations

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ADDITIONAL READING

The following list of documents will provide useful additional guidance and information on the topic of impact prediction:

INAP, 2008, Global Acid Rock Drainage Guide

USEPA, 1994, Acid rock drainage Prediction, Technical Document

Younger, P.L. and Sapsford, D.J., 2004, Evaluating the potential impact of opencast coal mining on water quality (Groundwater Regulations 1998) – An assessment framework for Scotland, Scottish Environment Protection Agency

Salmon, S.U. and Malmström, M.E., 2003, Geochemical processes in mill tailings deposits: modelling of groundwater composition, Applied Geochemistry

Banwart, S.A. and Malmström, M.E., 2001, Hydrochemical modelling for preliminary assessment of mine water pollution, Journal of Geochemical Exploration 74 (2001)

Maest, A.S. and Kuipers, J.R., Travers, C.L. and Atkins, D.A. 2005, Predicting water quality at hardrock mines – methods and models, uncertainties and state-of-the-art.

Price, W.A., 2005, List of potential information requirements in metal leaching and acid rock drainage assessment and mitigation work, MEND Report 5.10E

Pulles, W., Zhao, B., Usher, B.H. and Yibas, B. 2008. Evaluation and validation of geochemical prediction techniques for underground coal mines in the Highveld / Vryheid regions. WRC Report in preparation

Usher, B.H., Cruywagen, L-M., de Necker, E. and Hodgson, FDI., 2003. On-site and laboratory investigations of spoil in opencast collieries and the development of acid-base accounting procedures, WRC Report 1055/1/03

Usher, B.H., Cruywagen, L-M., de Necker, E. and Hodgson, FDI., 2003. Acid-base: Accounting techniques and evaluation (ABATE): Recommended methods for conducting and interpreting analytical geochemical assessments at opencast collieries in South Africa, WRC Report 1055/2/03

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- ASTM (1995) *Standard Guide for Design of Ground-Water Monitoring Systems in Karst and Fractured-Rock Aquifers*, ASTM Standard D5717-95. <http://www.astm.org>.
- ASTM (1995) *Standard Test Method for Measurement of Hydraulic Conductivity of Porous Material Using a Rigid-Wall, Compaction-Mold Permeameter*, ASTM Standard D5856-95. <http://www.astm.org>.
- ASTM (1995) *Standard Guide for Displaying Results of Chemical Analyses of Ground Water for Major Ions and Trace Elements-Diagrams Based on Analytical Calculations*. ASTM Standard D5877-95 (Reapproved 2000). <http://www.astm.org>.
- ASTM (1999) *Standard Guide for Selecting Surface Geophysical Techniques*, ASTM Standard D6429-99. <http://www.astm.org>.
- ASTM (2000) *Standard Test Methods for Measurement of Hydraulic Conductivity of Saturated Porous Materials Using a Flexible Wall Permeameter*, ASTM Standard D5048-00. <http://www.astm.org>.
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**APPENDIX A:
INTRODUCTION
TO PHYSICAL AND
CHEMICAL
PROCESSES
INVOLVED IN IMPACT
PREDICTION**

A1 GEOCHEMICAL ASPECTS

Although mining operations may result in a variety of water quality related impacts such as sulphate, metals, suspended solids, radionuclides, nitrogen compounds, oil and grease, etc., it is generally accepted that the primary water quality related problems are associated with acid rock drainage (ARD). For the purpose of this document, ARD related impacts are those that originate from one or more of the following processes:

- 1) Chemical and biological oxidation of sulphide minerals to release acidity, metals and sulphates
- 2) Dissolution of other metals and/or radionuclides by the acidity
- 3) In-situ neutralisation of the acidity by basic minerals to add contaminants such as calcium, magnesium and/or sodium

In the presence of sufficient basic minerals, the acidity released in the sulphide oxidation step may be completely neutralised and many of the metals, radionuclides and sulphates may be precipitated as secondary minerals. Contaminated mine water that originates from ARD may, therefore, either be acid or neutral, may contain high or low levels of metals and may have varying levels of sulphate, calcium, magnesium or sodium.

Various physical, chemical and biological driving forces, including the following, dictate the degree to which the mine water becomes contaminated:

- a) Physical and mineralogical nature and abundance of sulphide minerals
- b) Physical and mineralogical nature and abundance of neutralising minerals
- c) Water flow volumes, flow paths and contact times with sulphide and neutralising minerals
- d) Presence or absence of relevant catalysing bacteria
- e) Levels of oxygen

- f) Temperature
- g) Generation and transport of heat

The status of these driving forces is also dependent on the type of mine feature, with the following primary mining features being of interest:

- i) Coarse waste residue deposit (coarse discard dump, waste rock dump, spoils heap)
- ii) Fine waste residue deposit (slimes dam, coal slurry dam)
- iii) Opencast pit (need to distinguish between the following stages of a pit: open or backfilled pit; unflooded; partially flooded; flooded; operational; defunct)
- iv) Underground mine (need to distinguish between the following stages of an underground mine: unflooded; partially flooded; flooded; operational; defunct)

For the first three of the above mining features, an additional aspect that will affect the degree to which water contamination occurs is the presence or absence of a cover. For the purpose of this document, a cover is defined as a barrier that has an effect on the movement of water and/or oxygen into the mining feature. Such a cover may be a water cover, a soil cover, a vegetation cover, a layer of organic or other material that consumes oxygen or a combination of these.

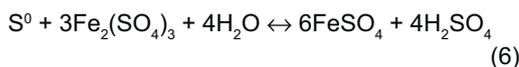
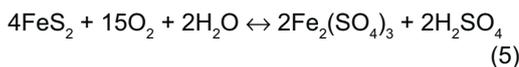
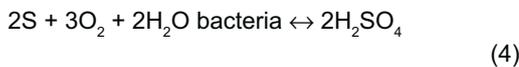
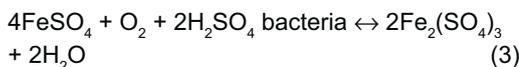
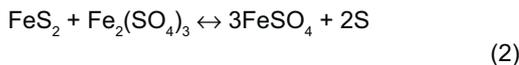
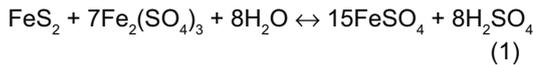
The ability to understand and then predict the manner in which the quality of water changes as it migrates through any of the mining features i to iv listed above requires an understanding of the following factors:

- Characteristics of the different mining features and how they affect the driving forces (a) to (g) as listed above - see Chapter 3.
- Characteristics and limitations of the different assessment techniques (laboratory analyses and predictive tools) - see Chapter 5

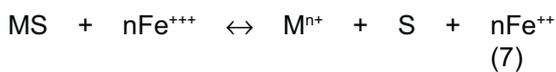
An overview of a number of these factors is given in Pulles, 1998.

A1.1 BASIC CHEMISTRY OF ARD GENERATION

The chemical reactions that describe the formation of ARD from pyrite (FeS₂) are shown below.



In the presence of a stable pH of 2.5 - 3.0, conditions may be created where the ferric iron itself can act as an oxidant of pyrite. Although the generated acid may dissolve some neutralising carbonate and oxide minerals, it is thought to have little effect on heavy mineral sulphides without the presence of the ferric iron. The action of the ferric iron on metal sulphides (including those of lead, copper, zinc and cadmium) is as follows (Mills, 1998a):



where MS = solid heavy mineral sulphide
 Fe⁺⁺⁺ = aqueous ferric iron ion
 Fe⁺⁺ = aqueous ferrous iron ion
 Mⁿ⁺ = aqueous heavy metal ion
 S = sulphur

It can be seen that the above reactions require the presence of the following four elements in order to occur:

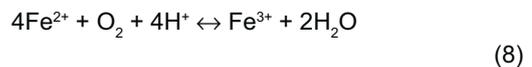
- sulphide mineral (e.g. FeS₂)
- oxygen (O₂)
- water (H₂O)
- bacteria (normally a strain of *Thiobacillus ferrooxidans*)

This is an important observation as **the successful assessment and prediction of ARD requires knowledge of all four of these elements (sulphides, oxygen, water and bacteria)**, while successful ARD management strategies will require the successful management or control of one or more of these elements. Additionally, all chemical reactions, including those responsible for ARD, are directly affected by temperature and this factor also needs to be understood and considered in impact prediction.

A1.2 BASIC MICROBIOLOGY OF ARD GENERATION

As shown in reactions 3 and 4 above, the sulphide oxidising bacteria play an important catalysing role in the generation of ARD. While these bacteria are generally considered to be *Thiobacillus ferrooxidans*, other species may also be found. These bacteria are termed autotrophs and obtain oxygen, nitrogen and carbon (CO₂) from the atmosphere. These bacteria have been reported to increase the sulphide oxidation rate by a factor of up to 1 000 000 (Evangelou & Zhang, 1995) although the real catalytic effect is likely to be significantly lower (10 to 1000 times) in most practical field situations.

A key reaction of the *T. ferrooxidans* bacteria is the oxidation of ferrous to ferric iron (Fe²⁺ to Fe³⁺) by way of the following reaction:



The ferric ion is a very powerful oxidant and can attack most metal sulphides in accordance with reaction 7, even at a Fe³⁺/Fe²⁺ ratio of 1: 1 000 000 (Dutrizac and MacDonald, 1974).

“Consideration of the effect of bacteria is most important in understanding the process of ARD generation. This is particularly so when “kinetic” tests are used to predict the rate of generation of ARD in the field. Only if the bacterial conditions of testwork are identical to or those in the field, can rates of ARD generation and/or metal solubilisation be taken from laboratory kinetic testwork and used to predict field behaviour with any degree of confidence” (Mills, 1998b).

A useful summary of the role of bacteria in ARD generation is given by Mills, 1998b while detailed reviews have been written by Brierley (1978) and Trudinger (1971).

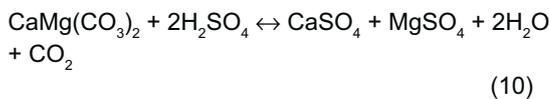
A1.3 BASIC CHEMISTRY OF ARD NEUTRALISATION

In addition to the generation of ARD, it is also necessary to consider the effect of the various neutralising minerals that occur within the mine ore body and that the ARD may come into contact with.

Calcite dissolution by sulphuric acid

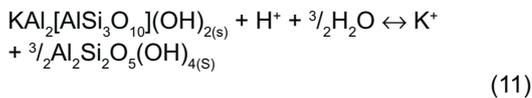


Dolomite dissolution by sulphuric acid

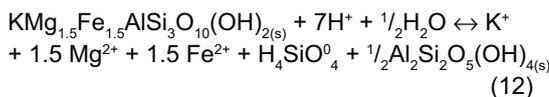


While the neutralising effects of the carbonate minerals are well established, the role of the various aluminosilicate minerals is not as clearly understood. However, depending on the pH of the ARD, the other minerals associated with the mine ore body may also make a very important contribution to the neutralisation of the ARD (Ritchie, 1994; Shaw & Mills, 1998). Examples of dissolution reactions that may occur for some of these minerals are:

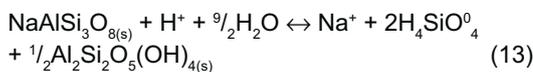
Muscovite dissolution



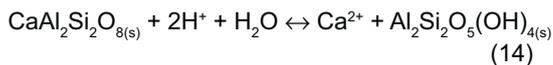
Biotite dissolution



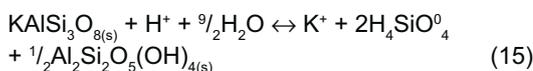
Albite dissolution



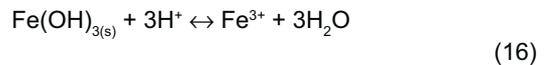
Anorthite dissolution



K-feldspar dissolution



Iron oxy-hydroxide dissolution



While there is some debate about the relative importance of the different minerals in the neutralisation of ARD, a table (Table A.1) of reactivity of the neutralising minerals (at pH 5) that has been widely quoted (e.g. Mills, 1998) is that of Sverdrup, 1990.

Table A.1: Relative Mineral Reactivity (after Sverdrup, 1990 and Mills, 1998)

Mineral Group	Typical Minerals	Relative Reactivity at pH 5
Dissolving	Calcite, aragonite, dolomite, magnesite, brucite	1.0
Fast weathering	Anorthite, nepheline, olivine, garnet, jadeite, spodumene, leucite, diopside, wollastonite	0.6
Intermediate weathering	epidote, zoisite, enstatite, augite, hypersthene, hedenbergite, hornblende, glaucophane, tremolite, actinolite, anthophyllite, serpentine, chrysotile, talc, chlorite, biotite	0.4
Slow weathering	albite, oligoclase, labradorite, montmorillonite, vermiculite, gibbsite, kaolinite	0.02
Very slow weathering	K-feldspars, muscovite	0.01
Inert	Quartz, rutile, zircon	0.004

It has been suggested that all minerals with a relative reactivity of 0.4 to 1.0 be considered as having practical neutralising capability in the field (Kwong, 1993a).

The effect of neutralisation of ARD containing heavy metals is that metals such as Fe^{3+} , Cu^{2+} , Zn^{2+} , Pb^{2+} and As^{3+} will eventually precipitate as secondary minerals in the carbonate or sulphate form and their hydrated and/or hydroxy-complex forms. These secondary minerals may re-dissolve at a later stage when the neutralising capacity has been exhausted or the rate of acid generation exceeds the rate of acid neutralisation, thereby releasing the metals back into the ARD.

Therefore, in addition to the important elements of sulphide mineral, oxygen, water bacteria and temperature, **knowledge of the neutralising mineral is also critical** in understanding the nature of ARD.

A1.4 THE ROLE OF ARD KINETICS

Knowledge of the rate (speed) at which a reaction takes place is critical to the understanding of the reactions and the final end products of the reaction, particularly when different processes such as ARD generation and ARD neutralisation need to be considered together in order to predict a resultant water quality. Chemical equations such as shown in equations 1 to 16 above are useful in showing the eventual end products under equilibrium conditions with infinite time available for the reactions to occur but do not indicate what will happen in real life at specific and unique sites.

In the real life ARD situation, the eventual end products are most often dictated by kinetic considerations.

For example, in a coarse waste rock or discard dump that has been assessed as having many times more neutralising potential than acid potential, the presence of rapid-draining preferential flow paths could result in a situation where the ARD passes by the neutralising minerals too quickly to allow effective neutralisation to occur. In addition, over a period of time, secondary minerals will tend to precipitate on the neutralising minerals, significantly reducing their reaction rates (e.g. by a factor of 5 for calcite). Table A.1 above also clearly indicates that different neutralising minerals have different reactivities and that at pH 5, calcite may be capable of neutralising sulphuric acid 2.5 times quicker than biotite. It is also known that the dissolution of simple carbonates such as calcite is predominantly equilibrium controlled but that the more complex carbonates and silicates are kinetically controlled (Li and Bernier, 1999). The practical importance of considering kinetics in assessing and predicting ARD generation has also been reported by Paktunc and Dave (1999) who reported that under unfavourable kinetic conditions, acidic drainage was generated even when more than 90% of the potentially available neutralising minerals still remained.

As another example, it has been established that the key rate limiting reaction in ARD generation is one where the ferrous iron (liberated during the oxidation of pyrite) is oxidised to ferric iron (reaction 8). This reaction is very slow and in the acidic pH region has a half time of 1000 days in the absence of bacteria. However, in the presence of *Thiobacillus ferrooxidans* this reaction proceeds at a reasonable rate.

As yet another example, observations generally conclude that decreasing particle size or increasing specific surface area lead to an increased leaching rate (Kargi and Weismann, 1984 and Laishley et al, 1986). This phenomenon has been quantified as a 1.5 times increase in leaching rate with a doubling of the surface area (Pugh, et al, 1984). The increased effect of large surface area holds for both chemical and bacterial oxidation of solid sulphides. Data so far suggest that conditions approach the optimum state when the particle size decreases below 37 - 44 μm (Tuovinen and Kelly, 1972). Knowledge of the particle size distribution is therefore critical in determining the rate at which ARD generation can occur.

According to Price (1999) the British Columbia (Canada) Ministry of Energy and Mines (BC MEM), who have made an extensive study of the assessment techniques for ARD, recognise the importance of kinetic considerations. *“Unlike practitioners who rely solely on static test results, BC MEM also requires a knowledge of the type, concentration and reactivity of minerals with neutralisation potential, likely exposure, physical and hydrological conditions, the rate of acid generation, along with kinetic data”.*

In a very extensive review of geochemical processes and geochemical models for the prediction of ARD from waste rock, the following conclusion was made: *“It is suggested that a model developed to predict water quality from waste rock piles should have a geochemical component that takes into account reaction kinetics”* (Perkins, et al, 1995).

A2 GEOHYDROLOGICAL ASPECTS

The rate and direction of groundwater contaminant migration from mine sites is dependant on a number of interrelated factors. There are two specific categories for subsurface contaminant migration, namely unsaturated and saturated flow.

The unsaturated zone is often considered to have an attenuating effect on the migration of contaminant plumes. According to Hamill and Bell (1986) this may be attributed in part to the rich bacterial populations that are present within soil. The relatively fine-grained nature of this zone promotes filtration of the suspended impurities. The organic and argillaceous material within this zone is also favourable for adsorption. According to these

authors, one of the most important characteristics of the unsaturated zone is the relatively slow vertical migration in contrast to the saturated zone where lateral migration is generally a lot faster. A more detailed description of the vertical percolation of water through the vadose zone is described in Driscoll (1986).

The impact of a potential pollution source therefore tends to be greater where the base of the source lies directly within the saturated zone such as an opencast pit or underground mine. Preferential pathways via cracks in the soil profile have a similar effect where the attenuation capacity of the unsaturated zone is short-circuited.

Once the contaminants enter the saturated zone, the migration is dominated by lateral migration. The rate of such contaminant migration is dependant on a number of factors such as:

- Groundwater Gradients
- Effective Porosity
- Hydraulic Conductivity
- Advection
- Dispersion and Diffusion
- Adsorption and Cation Exchange Capacity
- Biodegradation and Reactions

For the purpose of this document, the groundwater contamination migration will largely consider the inorganic constituents such as iron, sodium, calcium, magnesium and sulphates which constitute the bulk of the salt loading from mine sites in South Africa. An understanding of the geohydrological factors that influence groundwater contamination migration is essential as a basis for the prediction of future impacts on water quality. The manner in which the contaminants are initially solubilized are considered in the geochemical aspects of this document. The factors that influence the rate of groundwater contamination migration in the saturated zone are discussed in more detail as follows.

A2.1 PRINCIPLES OF ADVECTIVE GROUNDWATER CONTAMINANT MIGRATION

The primary driving force for groundwater flow and any associated contaminants is the groundwater gradient. Essentially groundwater flow occurs from higher to lower piezometric head. Contaminant migration that occurs as a result of this process is called advection.

Contaminants migrating in the groundwater regime are generally considered to be conservative where they are not retarded or attenuated by the host strata but rather migrate at the velocity of the subsurface water molecules. A number of interrelated factors govern the rate of advective contaminant transport which may be described by the following equation (after Driscoll, 1986):

$$V_a = \frac{K(h_1 - h_2)/L}{n} \quad (16)$$

where V_a = Actual water Velocity (m/day)
 K = (hydraulic conductivity m/day)
 $(h_1 - h_2)/L$ = Groundwater Gradient
 n = Effective Porosity

The influence of these parameters will be discussed in more detail later.

A2.2 GROUNDWATER GRADIENTS

According to Hamill and Bell (1986), water possesses three types of energy namely, potential energy attributable to its height, pressure energy due to its pressure and kinetic energy due to its velocity. Energy in water is usually expressed in terms of head. The head may be defined as the height to which water will rise within a monitoring borehole above a given datum. This height is often referred to as a piezometric level. Groundwater flow will therefore occur from a point of greater energy or piezometric head to a lower point of energy or piezometric head.

Groundwater gradients and groundwater contours are therefore an important indicator of the direction of groundwater flow. The steepness of the groundwater gradients also influences the velocity of the groundwater flow as will be discussed in later sections.

Groundwater gradients generally mimic the topography in fractured rock or secondary aquifers. However, exceptions to this rule include dolomitic and primary unconsolidated or sandy coastal aquifers. Caution should therefore be taken in the interpolation of the groundwater gradients in such aquifers.

Other factors that influence groundwater gradients or groundwater flow directions are geological structures such as faults and dykes. Such features may either act as groundwater conduits or preferential pathways for contaminant migration or may act as relatively impermeable barriers to groundwater flow such as the compartmentalization that has been observed in the dolomites.

A2.3 EFFECTIVE POROSITY

The porosity of any given volume of material is defined by the ratio of the volume of the pore space to the total volume of the material under consideration. Porosity which is usually expressed as a percentage provides an indication of how much water may be stored within a saturated medium. Porosity is also an important consideration for the rate of contaminant migration since groundwater flow will only occur through the actual interconnected pore space and not necessarily throughout the void volume. The actual or effectiveness of the interconnection between the pore spaces may be referred to as effective porosity.

A2.4 HYDRAULIC CONDUCTIVITY

Hydraulic conductivity is governed by the size and shape of the voids, the effectiveness of the interconnections between the voids and the physical properties of the fluid (Driscoll, 1986). The hydraulic conductivity is governed by the size of the pore spaces. For example if the pore spaces are small or constricted, then the hydraulic conductivity will be relatively low whereas the hydraulic conductivity will be higher where the pore spaces are larger. The hydraulic conductivity or ability to transmit water is obviously an important influence on the groundwater flow velocity.

A2.5 DISPERSION AND DIFFUSION

Dispersion is the result of variable flow velocities that are created by variable pore sizes and the degree of tortuosity or length of the flow path through which the water has to move. Dispersion results in a wider plume with decreasing concentration than would otherwise occur as a result of advection alone. According to Freeze and Cherry (1979), dispersion is stronger in the direction of flow (longitudinal dispersion) than in directions normal to the flow line (transverse dispersion).

Chemical dispersion or molecular diffusion may also occur where there is no groundwater flow. Since it is reasonable to assume that there will always be some degree of groundwater flow, this mechanism is relatively insignificant and will not be discussed further. The reader is referred to Freeze and Cherry (1979) for a more detailed discussion on these processes.

A2.6 GROUNDWATER CONTAMINATION RETARDATION PROCESSES

According to Freeze and Cherry (1979) there are a number of processes that retard the rate of groundwater

contaminant migration within the groundwater regime. These reactions may be sub-divided into six categories namely:

- Adsorption-desorption reactions
- Acid-base reactions
- Solution-precipitation reactions
- Oxidation-reduction reactions
- Ion-pairing or complexation (Aqueous Speciation)
- Microbial synthesis

Aspects of acid rock drainage such as the oxidation of pyrite and the neutralization potential of the host lithologies are discussed in detail in the geochemical component of the best practice guidelines. For a more detailed description on these processes, the reader is referred to Freeze and Cherry (1979) and Lloyd and Heathcote (1985). A synopsis of the chemical processes with reference to mining may also be found in Appendix A of the Water Research Commission report by Hodgson and Krantz (1998). Adsorption and ion pairing or complexation is described in more detail as follows:

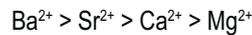
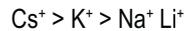
A2.6.1 Adsorption and Cation Exchange Capacity

According to Lloyd and Heathcote (1985), adsorbed contaminants are concentrated at the interface between the solid material and the solution. This is in contrast to adsorption that implies the permeation of the contaminants into the host material. Such processes are most prevalent where the host material is fine grained and has a large surface area such as clay minerals and iron and manganese oxides.

The process of adsorption is the result of unsatisfied valencies for atoms located at crystal surfaces. In oxides and silicates, the valency of the crystal surface is pH dependant. Clay minerals on the other hand have a permanent negative charge as a result of the substitution of silicates or magnesium or iron by aluminium.

The surfaces are surrounded by adsorbed counterions which balance the charge deficits. However, these ions are relatively mobile due to the weak electrostatic forces that bind them to the surface. These counterions which are commonly cations may be exchanged as a result. Hence the term cation exchange capacity which is higher for clays such as vermiculite and smectite than for kaolinites and illites.

In general cations are adsorbed according to the increasing hydrated ionic radius as follows:



A2.6.2 Ion Pairing or Complexation (Aqueous Speciation)

According to Hodgson and Krantz (1998), aqueous speciation is the formation of ion pairs through the association of individual cations and anions. Complex soluble ions are formed which influence the groundwater chemistry. For example depending on the water chemistry, only about 60% of sulphate occurs as SO_4^{2-} . According to these authors, the potential for gypsum precipitation is greatly reduced thereby allowing higher concentrations of sulphate and calcium to be mobile within the groundwater regime.

According to Freeze and Cherry (1979), the ability to predict the future concentrations of particularly trace metals will be dependant on the ability to predict the formation of complex ions in conjunction with redox reactions that may take place in the subsurface.

A2.6.3 Common Characteristics of Mine Related Groundwater Contamination Migration

Changing characteristics of the pollution sources are based on the geochemical reactions that take place within the mine workings or the mine waste residue. Once the leachate seeps into the unsaturated zone, it is subjected to a number of chemical reactions that have been described above. However, the rate of contaminant migration and the chemical signature of such a contaminant plume is dependant on the source of the contamination, the thickness of the unsaturated zone, the nature of the underlying lithologies and the hydrogeological characteristics of the underlying aquifers. Any groundwater contamination migration is therefore site specific and may be dominated by one or more of the available retardation processes. It is therefore impossible to describe the number of combinations and permutations that are available for the retardation of groundwater plumes from mining related activities. The potential for preferential flow is an important consideration for the rate of groundwater flow as well as the potential retardation reactions that are available.

Certain of the contaminants associated with mining activities are conservative such as sodium and

chloride. Sulphate is also relatively conservative since the estimated retardation for South African conditions is approximately 25%. However, trace metals and contaminants such as iron are not conservative and are therefore more difficult to predict in terms of the soluble complexes that may be formed. As discussed previously, the formation of such complexes will influence the ability to predict future groundwater concentrations of these elements.

A2.7 INFLUENCE OF MINE VOID STORAGE CAPACITY ON CONTAMINANT MIGRATION PREDICTIONS FROM DEFUNCT MINE WORKINGS

Aquifers surrounding active mining operations are commonly dewatered to varying degrees during the life of mine. Groundwater gradients generally stabilize towards the mining operations as a dewatering cone. Groundwater contamination migration within the sphere of influence of such a cone is therefore towards the active mine workings.

Once mining operations within a particular area have halted, the groundwater levels tend to recover to their ambient levels. This is provided that the defunct workings are isolated from other sections of the mine or surface decant points at lower elevations. During the groundwater level recovery, the dominant contaminant migration is still in the direction of the mine workings.

As the groundwater gradients recover to their ambient levels or a surface decant point is reached, the groundwater contamination migration tends to be away from the mine workings to the nearest surface water system or receptor in the receiving environment.

Clearly the rate of groundwater level recovery within defunct mine workings is an important consideration in the prediction of groundwater contaminant migration. One of the important aspects influencing groundwater recovery is the water storage capacity of the mine voids.

This storage capacity is strongly dependant on the mining method and the depth of the mining operations. For example bord and pillar mining will tend to have a larger storage capacity since the mine voids will tend to be open in contrast to total extraction or pillar robbing mining methods where the roof will form a goaf. The stopes of deep gold mines also tend to close after a period of time particularly where the reef is relatively flat as opposed to near vertical where closure of the stopes is not as pronounced.

An assessment of the mining method and the depth of mining is therefore essential to obtain an estimate of the mine void storage capacity.

A2.8 INFLUENCE OF GROUNDWATER INFLUX AND RECHARGE ON CONTAMINANT MIGRATION PREDICTIONS FROM DEFUNCT MINE WORKINGS

The volume of water ingress into defunct mine workings is another factor that influences the rate at which groundwater recovery occurs and the onset of offsite contamination migration. Water ingress into the mine workings consists of three components namely groundwater influx from the surrounding aquifers, surface water runoff and rainfall recharge.

The volume of groundwater influx will essentially be a function of the host aquifer's transmissivity and the depth of the groundwater dewatering cone surrounding the mine workings. Groundwater ingress into the defunct workings will be greatest during the initial stages of the groundwater level recovery when the groundwater gradients are at their steepest.

Rainfall recharge to the mine workings is likely to be relatively constant provided that there are no additional rehabilitation measures that may influence recharge following the cessation of mining. Other factors that will influence the volume of rainfall recharge and surface runoff into the mine workings will obviously be the wetter summer months as opposed to the drier winter months. It must be borne in mind that recharge often has a lag time between 2-4 months between the rainfall event and a variation in groundwater influx into the mine workings.

Any variation in the rate of groundwater recovery will depend on the proportion of the water ingress that is derived from groundwater influx as opposed to rainfall recharge or surface runoff. Estimates or direct measurement of these parameters are clearly important for the prediction of groundwater contaminant migration from affected mine sites.

A2.9 INFLUENCE OF THE LOCATION OF MINE RESIDUES ON THE PREDICTION OF CONTAMINANT MIGRATION PATTERNS

It is important to consider the location or siting of mine residue facilities on the prediction of the likely groundwater contaminant migration patterns from mine residue

wastes. There are four scenarios that may be considered for the siting of mine residue facilities namely:

- Waste residues located on the top of watersheds
- Waste residues located on hill slopes
- Waste residues located in river valleys
- Waste residues located on geological structures such as dykes and faults

The anticipated groundwater flow from these locations is shown graphically in Figure A1.

A2.9.1 Mine residues located at the top of watersheds

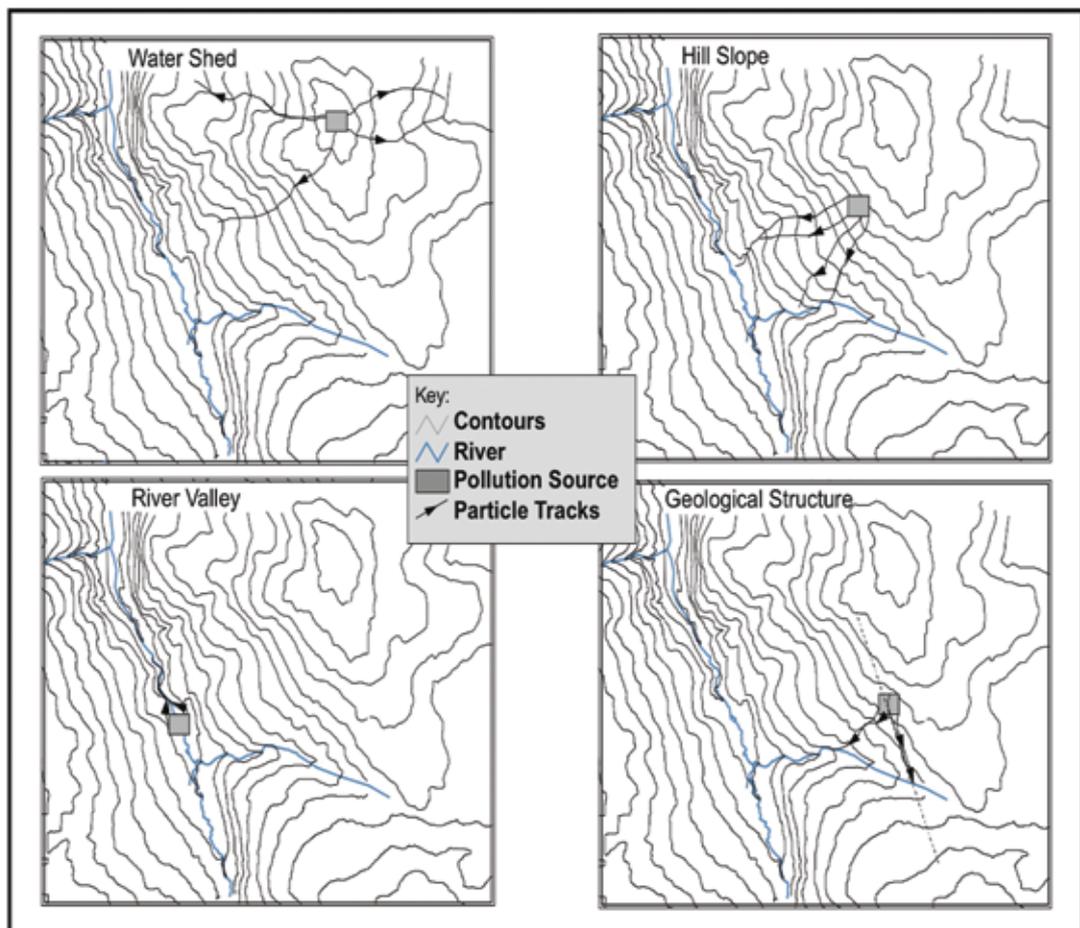
It is well understood that the groundwater gradients in South Africa generally mimic the topography. Obvious exceptions are dolomitic compartments and primary

coastal aquifers. Groundwater flow from geographical features such as watersheds will therefore occur in a number of directions. Although the depth to groundwater tends to be greater in the immediate vicinity of the watersheds, this benefit is offset against the relatively wider contaminant migration front that is associated with these features.

A worst case scenario is where the waste residue is located on a hilltop where the groundwater flow is likely to be radial from the contaminant source. Any pollution mitigation measures are difficult to implement under these circumstances since a greater contamination front has to be considered under these circumstances.

Ideally such locations should be avoided when siting mining waste residues.

Figure A1: Groundwater Flow Directions from Various Mine Residue Locations



A2.9.2 Mine residues located on the hill slopes

Groundwater flow patterns on the hill slopes generally flow in the direction of the nearest water course. The groundwater flow patterns tend to be fairly uniform under these circumstances with the contaminant migration occurring in one direction. Any contaminant mitigation measures are more easily implemented under these circumstances since the contaminant migration front is relatively limited in extent. The depth to groundwater is less than that of the watersheds. However, the decrease in attenuation capacity of the unsaturated zone is compensated by the restricted contamination plume.

Hillside slopes are generally the most favourable sites for mine waste residues since they represent the best compromise between the attenuation capacity of the unsaturated zone and the control of the groundwater contaminant migration from the waste residue footprint. Such a site should also lie outside the 1:100 year floodline. The gradient of the slope should also not be too excessive to avoid slope stability problems.

A2.9.3 Mine residues located in river valleys

Mining waste residues have traditionally been located in river valleys given the relative ease of constructing retaining walls etc. However, there are a number of issues that are involved in such sites namely:

- The waste residue is located within the floodlines
- Relatively clean groundwater that converges in the vicinity of the river system will become contaminated as it comes into contact with the waste residue
- There is often no unsaturated zone and therefore no attenuation capacity within the immediate vicinity of the river systems
- The accumulated thickness of unconsolidated material within the river course may prevent the effective interception of any groundwater contamination migration by means of traditional methods such as cut-off trenches.

Although the contamination plumes may be diluted by the convergence of clean groundwater at the base of the river course, the contaminants still contribute to the contaminant load of the river system. Sites within water courses are not considered to be acceptable and are restricted in terms of GN704.

A2.9.4 Mine Residues located on geological structures

The primary porosity of the majority aquifers within South Africa is generally negligible. Groundwater therefore primarily flows within secondary structures such as faults, joints and the contact zones of dolerite intrusions such as dykes and sills. These geological structures are often targeted for water supply wells due to their greater permeability and storage capacity.

Mine waste residues that are located on geological dykes and faults are therefore often associated with greater contamination plumes than would otherwise be the case. A notable example is that of the Orange Free State where contaminant plumes have migrated some 100m from gold tailings dams over the last 25 years. However, the contamination plume associated with a tailings dam that was located on the contact zone of a dolerite sill has migrated some 3km over the same time period.

It is therefore important that any geological structures are identified at the outset to reduce the impact of the mine waste residue on the environment. Any potential geological features also need to be identified for existing facilities so that more realistic predictions of any groundwater contamination migration may be made.

A2.10 ACCURACY OF GROUNDWATER CONTAMINATION PREDICTIONS FROM MINE SITES

The accuracy of any contaminant migration predictions from mine sites will be dependant on how representative the monitoring and field measurements are of the immediate physical environment with respect to the above factors that influence the rate of contaminant migration.

Given that geophysical surveys and the drilling of boreholes are expensive, the number of observation or sampling points in any geohydrological investigation are limited. For this reason, the interpretation of the available field data to formulate a groundwater conceptual model is critically important to the successful prediction of any mine related impacts. Such work should only be undertaken by an appropriately qualified person.

APPENDIX B: IMPACT PREDICTION TOOLS AND PROCEDURES

This Appendix describes the different tools and techniques and their key characteristics, strengths and limitations. This BPG does not aim to discuss each technique in detail and references are given to enable interested persons to obtain additional details.

B1 QUALITY ASSURANCE / QUALITY CONTROL

The Quality Assurance/Quality Control (QA/QC) process is an ongoing one that must be integrated into the whole assessment and pollution prediction programme. The process starts **before** the first samples are taken and continues after the predictions have been made. Elements of the QA/QC process that can be distinguished are the following:

- Development of conceptual model and identification of tools to be used
- Development of sampling programme (for field, static and kinetic tests)
- Sampling, preservation, preparation and chain-of-custody
- Duplicates, standards and use of alternative analytical techniques
- Use of reputable laboratories to undertake analytical programme
- Databases and statistical interpretation
- Use of suitably qualified persons to undertake mathematical modelling and use of appropriate models for the problem
- External review
- Monitoring and model calibration
- Documentation of QA/QC programme

Each of these QA/QC aspects is briefly discussed below. A good review of quality assurance/quality control issues for ARD studies is given by Downing & Mills (1998) and readers of this BPG are referred to this source for additional detail and additional references on QA/QC issues.

B1.1 DEVELOPMENT OF CONCEPTUAL MODELS AND IDENTIFICATION OF TOOLS TO BE USED

The very first step that needs to be undertaken before any sampling or analytical programme is embarked upon is to define the following:

- Understand and define the mining scenario that needs to be assessed.
- Define the important chemical, physical and microbiological factors that need to be considered when making the assessment and predicting future pollution.
- Define the key questions that need to be answered by the assessment.

On the basis of the above, a preliminary conceptual model of the mining scenario to be assessed needs to be developed that clearly incorporates the relevant issues. Based on this conceptual model and an understanding of the capabilities and limitations of the available tools, the appropriate tools that should be used for the pollution prediction assessment need to be defined (it should be recognised that the preliminary conceptual model may change on the basis of collected information). The different prediction models have different data requirements and these must then be considered when the sampling programme is developed.

This step is one of the most critical elements in the QA/QC process as the wrong conceptual model and the wrong choice of tools will almost certainly result in an inappropriate sampling programme and the generation of inappropriate data.

B1.2 DEVELOPMENT OF SAMPLING PROGRAMME

The development of a good and appropriate sampling programme is considered absolutely critical to any pollution prediction assessment. Due to the natural variability that can be expected in the parameters to be measured and the sheer spatial extent of most mining scenarios, it is very important to ensure that the sampling programme is statistically designed to ensure that it yields data that is truly representative. The sampling programme also needs to be flexible and should be reviewed on the basis of collected data to ensure that the original concepts that dictated the first sampling programme remain valid. While there are no strict guidelines that categorically define an appropriate sampling programme, professional best judgement must be used and there must be an ability to defend the programme against external and independent review. Later on in this Appendix and in **BPG G3: Water Monitoring Systems** further detail is given on the actual taking and handling of samples.

When developing the sampling programme and making use of North American guidelines and references it is important to also understand the differences in the assessment objectives between North American and South African sites. While there are obvious climatological and ore body differences, it is **particularly important to take note of the different regulatory requirements**. In North America (USA and Canada) and other countries, the primary regulatory focus is normally on the generation of acidity and the release of toxic heavy metals, while in South Africa there is also generally a regulatory concern with dissolved salts such as sulphate. An assessment and sampling programme that is geared to defining the risk of the onset of acidic conditions, is very different to one that is geared to predicting long-term water qualities in terms of dissolved contaminants such as sulphate, which may arise without the onset of acidic conditions.

The sampling programme must, in addition to consideration of the parameters to be measured, also take account of the following factors:

- Life-cycle status of the mine
- Geological and value continuity
- Sampling techniques

B1.2.1 Life-cycle status of the mine

It is probable that different sampling programmes will be developed for mines depending on the stage in the life cycle of the mine. For a baseline assessment prior to

the commencement of mining operations, the sampling techniques will generally be less invasive and of a more environmental nature. Extensive sampling of the ore body and waste material to be generated is generally not possible at this stage.

During the exploration stage and the development of a premining plan, it will be possible to obtain extensive samples of the material to be mined. It is important to include the need for samples for waste characterisation, pollution prediction and management into the overall exploration phase. It is common for mines to primarily consider the ore body from an economic exploitation perspective at this stage and drill core samples are often entirely consumed for this purpose unless the waste characterisation aspect is fully incorporated into the exploration phase.

During the operational phase of a mine, it is generally possible to obtain the widest range of data. It is important, at this stage in a mine's life, to already consider the closure and post-closure data requirements and to obtain such data while all the relevant areas of the mine are accessible.

At the point of mine closure, it is often found that many areas of the mine are inaccessible and that key data cannot easily be collected. This situation should be avoided as far as possible by ensuring that appropriate data collection occurs during the operational phase of the mine. For mines that are already at the closure stage and where it is not possible to access all the relevant areas of the mine and where existing appropriate data are not available, special sampling and assessment programmes may be required. The data limitations should also be considered when deciding which mathematical models to use in the pollution prediction exercise and special consideration should be given to undertaking probabilistic modelling in order to define the uncertainty in the post-closure phase of the mine.

B1.2.2 Geological and value continuity

Continuity is important in any assessment programme and can be considered to comprise of two parts (Downing, 1999), viz. geological continuity and value continuity. Geological continuity refers to features that control mineralisation, while value continuity refers to a value or measure such as acid potential, neutralisation potential, etc. and the manner in which this variable occurs spatially. A well designed sampling programme must reflect both these aspects. One example of a particular assessment technique that addresses continuity is block modelling

which can also be used for ARD studies (Downing and Giroux, 1993, 1998).

B1.3 SAMPLING, PRESERVATION, PREPARATION AND CHAIN-OF-CUSTODY

Aspects to consider when taking water samples are covered in **BPG G3: Water Monitoring Systems**. With regard to geochemical sampling, it is important that all the relevant lithologies should be sampled. While this is generally understood for waste deposits, for open cast mines this means that overburden, waste rock and tailings (slurry) should be separately sampled. For underground mines, it will generally require the inclusion of ore body, roof and floor material as well as secondary minerals that have been deposited within the mine.

Each type of sample should be clearly marked as to the material sampled and the precise location (using GPS or some other survey system) and date of the sample - where possible, photographs of the sample site should be taken. The size of the sample to be taken will depend on the heterogeneity of the sample and the analytical techniques that are to be applied. For static geochemical analytical techniques a sample size of between 1 and 5 kg is generally adequate, while more may be required for kinetic tests (depending on the apparatus to be used).

Samples that are taken for subsequent geochemical assessment must be prevented from undergoing further oxidation by being stored in sealed containers, while wet samples should be dried (at < 40 °C) as soon as possible. Ideally samples should be stored under nitrogen to prevent both chemical and microbiological oxidation processes from occurring.

According to Downing (1999) the following different types of samples could be incorporated into the sampling programme: point samples (single or composite); linear samples (e.g. drill cores, channel samples); panel samples; or bulk samples.

The chain of custody of each sample should be clearly maintained and documented from the point when the sample is taken, through shipping, analysis and storage. The purpose of maintaining a clear chain of custody is to ensure that sampling and handling protocols are adhered to and that inappropriate handling or contamination risks are minimised. The chain of custody documentation must be kept in order that any subsequent queries

regarding the reliability of data can be satisfactorily answered. According to Downing and Mills (1998), this documentation should include the following:

- 1) The date, time and sampling protocol for the original sample.
- 2) The method, duration and location of any sample storage.
- 3) A detailed record of any physical or chemical treatment of the sample, including drying, crushing, grinding, screening, splitting and washing.
- 4) A record of all personnel who have handled the sample, including time and place.
- 5) Record of all disposals of sample parts, fractions and splits.

B1.4 DUPLICATES, STANDARDS AND USE OF ALTERNATIVE ANALYTICAL TECHNIQUES

The QA/QC programme should make provision for the following checks:

- duplicate samples within and between laboratories
- analysis of standard materials (e.g. CANMET NBM-1 ABA Standard)
- use of alternative analytical methods to cross-check the chosen methods (e.g. it is recommended that approximately 5 - 10 % of total samples be cross-checked for the various analyses that are undertaken)

B1.5 USE OF REPUTABLE LABORATORIES TO UNDERTAKE ANALYTICAL PROGRAMME

It is obvious that the level of confidence of pollution predictions is directly related to the level of confidence of the input data. It is, therefore, imperative that considerable care be given to selecting reputable laboratories to undertake the analytical programme. In this context, it is probable that different laboratories will be selected to undertake different analyses. According to Calow (1991) as referenced by Downing & Mills (1998) common laboratory errors can be classed as follows:

- Incorrect identification of samples
- Contamination of samples
- Improper or inappropriate sample preparation
- Inaccuracy of sample weights or volumes

- Improper or inappropriate sample dissolution/treatment
- Chemical and physical interference
- Improper or inappropriate instrumentation / inaccurate measurement
- Calculation errors
- Incorrect data handling / reporting

It is also recommended that the selected laboratory be a participant in one or more of the inter-laboratory comparison programmes for the analyses of interest. For organisations that will be requiring a large amount of geochemical analytical work to be undertaken, it could be worthwhile to undertake an independent audit of the laboratory's procedures to ensure confidence in the reported data.

Finally, experience has shown that it is also important for the geochemical practitioner to be intimately familiar with the different analytical techniques in order to be able to ensure that the laboratories undertake the correct procedures. Clear specification of detection limits and required accuracy should also be made.

B1.6 DATABASES AND STATISTICAL INTERPRETATION

Large amounts of data can be generated in geochemical, geohydrological and pollution prediction exercises and the development and ongoing maintenance of suitable computerised databases is an essential part of the overall QA/QC programme. For large mine site assessments, particularly where actual mining operations (underground or opencast) are involved, it is considered essential to enter the data into a suitable Geographic Information System (GIS) in order to support the spatial assessments and spatial data manipulations that will inevitably be required. These aspects are also discussed in Section B2.2 of this BPG and in **BPG G3: Water Monitoring Systems**.

It is important to set up QA/QC checks at various stages in the data handling process, including the following:

- Review of all data by a technically competent person before it is entered into the database in order to check its integrity.
- Regular review of complete data sets to check for outliers and to confirm the validity of these or to confirm a motivation to ignore the data in subsequent assessments. In this instance it is important to recognise that outliers may represent natural variability and may not necessarily be errors.

- Ensure regular backups of computerised databases.
- Undertake regular manual checks of computerised data manipulation routines in order to check their accuracy - there is all too often a tendency to automatically assume the correctness of calculations undertaken by computers.

The analysis and statistical interpretation of data is an important component of the QA/QC process in order to identify questionable data. There are a wide range of statistical interpretation options that could be undertaken, depending on the questions that need to be answered. There are a number of good statistical computer packages that can assist in the plotting and interpretation of data. Examples of some relevant statistical interpretations of geochemical data are given in Downing and Mills (1998).

B1.7 USE OF SUITABLY QUALIFIED PERSONS

The prediction of pollution from mining sites is a very complex exercise that requires the integration of a number of specialist skills such as geochemistry, mineralogy, petrology, microbiology, civil engineering, geohydrology, hydrology, soil science, and statistics and it is highly unlikely that any single person could claim to adequately cover all these required disciplines. Experience has shown, therefore, that the use of a team approach that combines the key required skills is the only route to successful geochemical and pollution assessments / predictions.

While it can confidently be stated that a range of tertiary qualifications are a prerequisite to undertake a complex assessment such as is covered in this BPG, there are problems in rigidly defining the appropriate qualifications. For example, it is possible to obtain a 1-year post-graduate degree in geochemistry without any undergraduate qualifications or training in chemistry or geology, leading to persons with perhaps less geochemical assessment capability than a chemist or mineralogist. Similar discrepancies arise with post-graduate training of geohydrologists. On the other hand, it is also possible for persons that have no undergraduate training in some of the key disciplines to acquire the necessary skills through appropriate work experience.

It is proposed that for persons to be considered suitably qualified to undertake geochemical assessments, that they should be able to demonstrate a capability to clearly and scientifically defend their methodologies, assessments and conclusions on the basis of best practice as defined

in this BPG. It is also proposed that for assessments that have significant potential environmental impacts, that the suitably qualified persons should willingly and voluntarily submit their assessments to independent external review as discussed in Chapter 8. In addition to the key specialists involved in a geochemical / geohydrological assessment, it is also essential to ensure that all project staff, particularly field samplers and data collectors are suitably trained to undertake the tasks entrusted to them. It is proposed that the specialists undertaking an impact prediction should attach a signed letter to the project proposal and ultimately to the final project report, identifying the areas of specialist skills required for the project and confirming their competence in undertaking such a study.

B1.8 INDEPENDENT EXTERNAL REVIEW

Due to the complexity of typical pollution prediction assessments, the range of skills required, the need to successfully integrate these skills, the unavoidable uncertainties encountered in such assessments and the potential environmental and financial impact of wrong assessments, it is considered good practice to incorporate independent external review into the assessment.

Due to the complexity of the assessment for very large mining sites, it is proposed that such external review be integrated into the project with review occurring on an ongoing basis and not only at the end of the assessment when it is difficult, if not impossible, to act upon the review recommendations. A suitably qualified reviewer should be able to demonstrate previous exposure and involvement in similar assessments and should not be in the permanent employ of the mine site owner or the specialists undertaking the assessment. More detail on the proposed review process is provided in Chapter 8.

B1.9 MONITORING AND MODEL CALIBRATION

As with any modelling exercise, there will always be uncertainty in the predictions made with regard to future pollution from mining sites. It is considered essential and non-negotiable that any pollution prediction exercise be followed up with an appropriate monitoring exercise with subsequent review and calibration of the models, particularly where there is significant variance between predictions and actual data. Such a monitoring programme should not only allow for monitoring of the end result (e.g. volume and quality of water) but should also include monitoring of key variables incorporated

into the models. Sensitivity analyses at the modelling stage should be undertaken in order to identify the key variables that should be monitored.

An alternative approach is for the mine to construct field scale plots and test sites to demonstrate the performance of various management options during the life cycle of the mine and to then use this data to calibrate and validate the models at the earliest opportunity.

If the monitoring programme generates data that are completely inconsistent with the predictions that were made, the reason for the discrepancy will need to be determined. In the first instance, the monitoring data should be used to rigorously interrogate the conceptual model and to confirm its suitability. If the conceptual model does hold up, then the various assumptions and data sets will need to be interrogated to determine where the errors lie and to then undertake additional modelling with revised data sets that do correlate with the monitoring results.

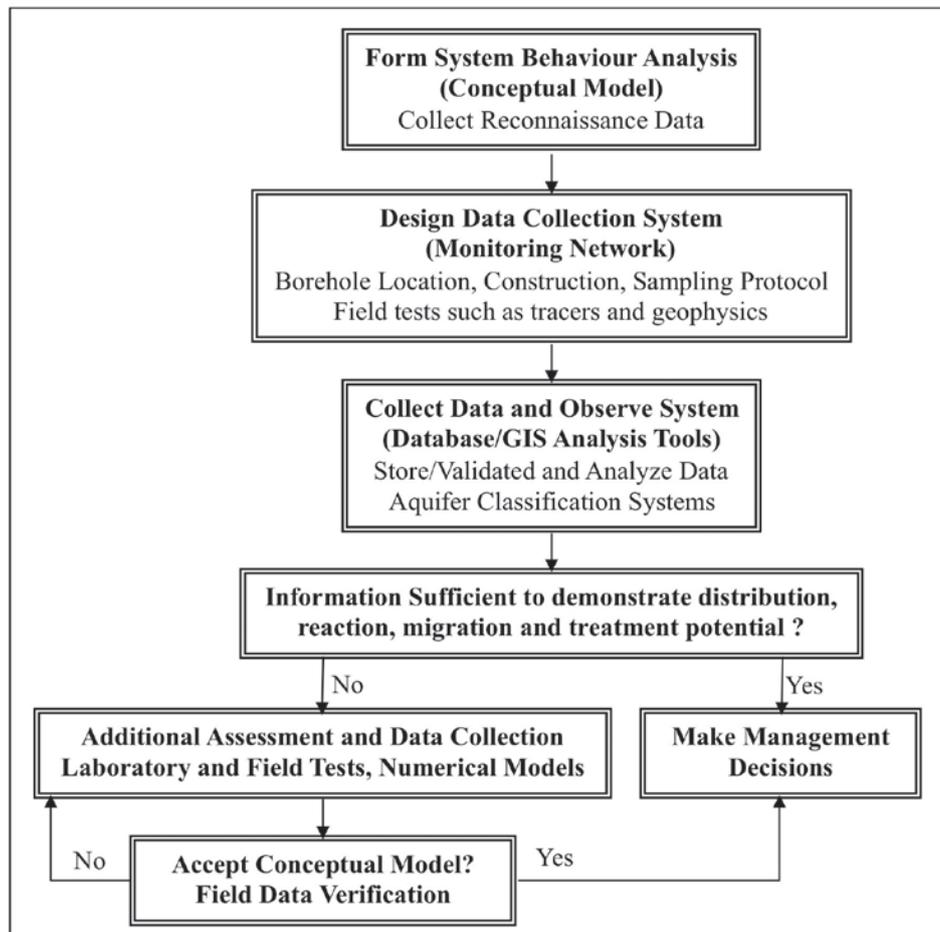
B1.10 DOCUMENTATION OF QA/QC PROGRAMME

The complete QA/QC programme, covering all the aspects discussed above should be clearly documented and available for review at any stage of the assessment.

B2 GEOHYDROLOGICAL TOOLS AND TECHNIQUES

Geohydrological tools are used to geohydrologically characterize the mining site as well as determine any potential remedial measures that may be required in terms of groundwater flow/pumping levels and contamination migration from potential sources. Although these tools vary in sophistication, they all aim to determine or utilize the aquifer parameters, piezometric levels and water qualities in the vicinity of the site. Site characterization of a mine site is an iterative process as indicated in Figure B1 (Adapted from the US EPA Site Characterization Seminar Report, 1991)

Figure B1: Mine Site Characterization flow chart



It can be seen from Figure B1 that the available geohydrological tools provide information and field data during the following stages of the site characterization namely:

- Conceptual Modelling
- Data Collection or Monitoring Network Design
- Data Analysis and Modelling

The available geohydrological tools are discussed in terms of each of the above processes on the basis of currently documented guidelines and procedures.

B2.1 CONCEPTUAL MODELLING

According to Anderson and Woessner (1992), the purpose of the conceptual model is to "simplify the field problem and organize the associated field data so that the system may be organized more readily". Conceptual

modelling is also a useful framework for the collection of field data. Such a conceptual model is usually refined following the processing of the field observations from preliminary investigations. Since conceptual modelling forms the foundation for all subsequent investigations and predictions, it is the single most important activity that can be undertaken by a geohydrologist. It is often the incorrect conceptual modelling or lack thereof that leads to incorrect data collection and predictions.

The foundation for groundwater conceptual modelling for Southern African is the determination of whether the aquifer is primary (granular), fractured or karstic. The differences between these aquifers as described in the ASTM standard D5717-95 are presented in Table B1

Table B1: Comparison between Granular, Fractured and Karstic Aquifers (after ASTM D5717-95).

Aquifer Characteristics	Aquifer Type		
	Granular	Fractured	Karstic
Effective Porosity	Mostly primary, Through intergranular pores	Mostly secondary, through joints, fractures, and bedding plane partings	Mostly tertiary (secondary porosity modified by dissolution); through pores, bedding planes, fractures, conduits, and caves
Isotropy	More isotropic	Probably anisotropic	Highly anisotropic
Homogeneity	More homogeneous	Less homogeneous	Non-homogeneous
Flow	Slow, laminar	Possibly rapid and possibly turbulent	Likely rapid and likely turbulent
Flow Predictions	Darcy's law usually applies	Darcy's law may not apply	Darcy's law rarely applies
Storage	Within saturated zone	Within saturated zone	Within both saturated zone and epikarst
Recharge	Dispersed	Primarily dispersed, with some point recharge	Ranges from almost completely dispersed- to almost completely point-recharge
Temporal Head Variation	Minimal variation	Moderate variation	Moderate to extreme variation
Temporal Water Chemistry Variation	Minimal variation	Minimal to moderate Variation	Moderate to extreme variation

According to the ASTM D5717-95 guideline, three conceptual models of groundwater flow may be used to characterize fractured and karstic aquifers as follows:

- **Continuum** – This model assumes that the aquifer approximates porous media at some workable scale. The properties of the individual fractures and conduits are not as important as large volumes of the aquifer material. Field and laboratory tests for porous media are applicable here. (Model not applicable where flow occurs predominantly through widely spaced fractures or conduits).
- **Discrete** – This model assumes that flow occurs predominantly along fractures and conduits. The contribution of the matrix is unimportant in this model. Hydraulic characteristics of the fractures are used to define the groundwater regime. (Great contrast between hydraulic characteristics of fractures and matrix).
- **Dual Porosity** – This model lies somewhere between the continuum and discrete models where one attempts to characterize both the fractures and matrix. (Matrix

is relatively permeable but discrete zones of higher conductivity and preferential pathways do exist).

The application of the specific conceptual model is dependant on the scale of the investigation. ASTM guideline D5717-95 lists a checklist to classify groundwater regimes within one of the above conceptual models as follows:

- **Ratio of Fracture Scale to Site Scale** – for porous medium equivalent aquifers the vertical and horizontal fractures should be numerous, the distances between the fractures should be orders of magnitude less than the site under investigation and fractures should show appreciable interconnection.
- **Hydraulic Conductivity Distribution** – the distribution in porous medium equivalent aquifers tend to be log normal. Bimodal or polymodal distributions of hydraulic conductivity are unlikely to be representative of porous media. It is possible to have log-normal hydraulic conductivity distributions for non-porous media since boreholes are often sited preferentially on fractures. However, hydraulic conductivity values

on open boreholes do not necessarily reflect the actual variation for the site.

- *Water Table Configuration* - Porous media generally have smooth groundwater contours in contrast to the step like patterns that may be associated with fractured rock aquifers.
- *Pump Test Responses* – Drawdown in the observation boreholes should increase linearly with increases in abstraction rates. Time-drawdown curves for monitoring boreholes located at a similar distance from the pumping well should show a similar shape and have no sharp inflexion points. Plotted drawdown cones around the pumping well should be circular. Elongated drawdown cones are not representative of porous media.
- *Variations in Water Chemistry* – Large spatial and temporal variations in groundwater chemistry tend to be characteristic of fractured or karstic aquifers due to migration through discrete conduits.
- *Variations in Hydraulic Head* – Rapid variation in response to recharge may be characteristic of fractured or karstic aquifers.
- *Borehole Logging* – Fractures or solution cavities are often observed during drilling of the monitoring boreholes.

Once the fundamental conceptual model for the site has been selected, the following data is collected to characterize the flow regime further. A list of the data required for the prediction of groundwater flow is defined in the US EPA Site Characterization Seminar Report (1991) as follows:

Physical Framework

- Hydrogeologic map showing areal extent and boundaries of aquifer.
- Topographic map showing surface-water bodies.
- Water-table, bedrock-configuration, and saturated-thickness maps.
- Hydraulic conductivity map showing aquifer and boundaries.
- Hydraulic conductivity and specific storage map of confining bed.
- Map showing variation in storage coefficient of aquifer.
- Relation of stream and aquifer (hydraulic connection).

Stresses on System

- Type and extent of recharge areas (irrigated areas, recharge basins, recharge wells, impoundments, spills, tank leaks, etc.).
- Surface water diversions.
- Ground-water pumpage (distributed in time and space).
- Stream flow (distributed in time and space).
- Precipitation and evapotranspiration.

Observable Responses

- Water levels as a function of time and position.

Other Factors

- Economic information about water supply.
- Legal and administrative rules.
- Environmental factors.
- Planned changes in water and land use

The data that is pertinent to the prediction of groundwater contaminant migration is as follows:

Physical Framework

- Estimates of the parameters that comprise hydrodynamic dispersion.
- Effective porosity distribution.
- Information on natural (background) concentration
- Distribution (water quality) in the aquifer.
- Estimates of fluid density variations and relationship of density to concentration (most important where contaminant is salt water or results in significantly higher concentration of total dissolved solids compared to the natural aquifer or where there are significant temperature differences between the contaminant plume and the natural aquifer).

Stresses on System

- Sources and strengths of pollutants,

Chemical/Biological Framework

- Mineralogy media matrix.
- Organic content of media matrix.
- Ground-water temperature.
- Solute properties.
- Major ion chemistry.
- Minor ion chemistry.
- Eh-pH environment.

Observable Responses

- Areal and temporal distribution of water quality in the aquifer.
- Stream flow quality (distribution in time and space)

Any data gaps in the above list that have not been determined from the reconnaissance search are listed as an objective for the monitoring network design. The reader is referred to Anderson and Woessner (1992), the US EPA Site Characterization Report Seminar (1991) and ASTM standard D5717-95 for a more detailed description of groundwater conceptual modelling.

B2.2 DATA COLLECTION OR MONITORING NETWORK DESIGN

The groundwater-monitoring network forms the basis for the field observations and measurements during a geohydrological investigation. Not only does the monitoring network provide information regarding the spatial variation in aquifer parameters in the immediate vicinity of the site, but it also delineates variations in the water quality. Monitoring is also a key component of the process of validation and calibration of predictions. According to Barcelona et. al. (1985), due care to ensure the collection of unbiased, precise hydrologic and chemical data should be exercised from the outset in all monitoring efforts. There are a number of aspects that need to be considered during the design, installation and data collection from such a system namely:

- The location and siting of the monitoring boreholes
- The drilling and construction of the individual monitoring boreholes
- Determination of Aquifer Parameters
- Groundwater Sampling

There are three types of monitoring boreholes that may be utilized during the installation of a groundwater-monitoring network. These are:

- An upgradient or ambient water quality monitoring well to establish the aquifer water quality deviations as a result of groundwater contamination migration
- A source monitoring borehole that is drilled immediately downgradient from the pollution source to establish the current groundwater quality due to the contamination source
- A calibration or predictive monitoring borehole that monitors the future or predicted water quality over time before it reaches a receptor.

According to the US EPA 1999 site monitoring plan for landfill sites, it is recommended that a minimum of three monitoring boreholes be drilled for each groundwater system consisting of one upgradient and two downgradient holes. Monitoring boreholes immediately adjacent to potential contaminant sources are also required. However, it is proposed that the specification of the number and location of monitoring boreholes be defined by a suitably qualified specialist who is capable of considering site-specific factors within his decision-making process.

The EPA document also recommends one monitoring borehole for each groundwater flow path between the potential contamination source and groundwater receptors. However, the separation between monitoring boreholes is a site specific consideration. It is clear from the above criteria that the layout of monitoring boreholes is dependant on the number of sources, groundwater receptors and potential groundwater flow paths. Regional groundwater flow paths may be determined from air borne geophysical techniques such as gravity for karstic aquifers and from structural maps while preliminary plumes may be delineated by techniques such as thermal imagery. However, the use of such relatively expensive techniques should be weighed against the objectives of the site investigation.

According to the ASTM standard D5717-95, significant fractures generally decrease with depth, which affects the depth of the flow regime within most fractured rock aquifers. According to the ASTM guideline, groundwater gradients generally mimic the topography under these conditions. In the absence of known geological structures that may influence the groundwater flow paths across the site, it is recommended that the preliminary location of the monitoring boreholes are determined by means of particle tracking on based on Bayesian interpolated water levels. This latter process tends to optimize the layout of the monitoring borehole network rather than drilling on a grid pattern that does not take the groundwater flow patterns into account.

Once the layout of the monitoring boreholes is established, the detailed siting of the individual boreholes takes place. Groundwater flow in South Africa is primarily associated with secondary geological structures such as faults and joints. According to the ASTM standard D5717-95, properly sited boreholes are more likely to receive contaminants from the site than randomly sited boreholes in a fractured environment. It is therefore imperative that such structures are located during the detailed siting

of the monitoring wells. These structures are most commonly identified by aerial photo interpretation and detailed geophysical surveys. The close proximity of infrastructure is no longer an excuse not to undertake a geophysical survey given the modern techniques that are currently available.

The type of geophysical survey that is undertaken for the detailed siting of the monitoring boreholes is dependant on the geohydrological setting. Groundwater occurrences are often associated with the base of weathering, geological structures such as faults and joints, solution cavities in dolomites and the contact zones of dolerite intrusions. The purpose of a geophysical survey is therefore primarily three-fold:

- To establish the base of weathering (e.g. seismic and gravity)
- To identify geological structures and dolerite contact zones (e.g. EM34 and Magnetometer)
- To delineate contamination plumes where possible (e.g. EM34 and electrical resistivity)

It is important to note that the geophysical survey and subsequent interpretation of the results must be undertaken by a suitably qualified person.

There are a number of geohydrological parameters that need to be determined or estimated before predictions can be made regarding groundwater contaminant migration. These aquifer parameters include the hydraulic conductivity or permeability, effective porosity and dispersion. The determination of these aquifer parameters must be undertaken by a suitably qualified specialist.

Groundwater sampling is important since it determines the variation in groundwater levels and quality in the immediate vicinity of mining operations. The background or ambient groundwater quality is also established simultaneously. The reader of this BPG is referred to a recent Water Research Commission Report prepared by Weaver et al (2007) for recommendations applicable to South Africa for groundwater sampling. It is important to ensure that the groundwater samples are representative and that they do in fact measure the water quality along the determined pathway.

B2.3 DATA ANALYSIS AND MODELLING

The data collected from the monitoring network is stored and subsequently analysed to validate the conceptual model regarding the site. The following

tools are commonly utilized during this phase of the geohydrological investigation:

- Database/GIS Data Storage and Analysis System
- Aquifer Vulnerability and Classification Schemes
- Analytical Techniques
- Graphical Techniques
- Numerical Modelling

These various components will be discussed in more detail as follows:

B2.3.1 Database/GIS Data Storage and Analysis System

Large volumes of field data are generated during most site investigations including temporal water level and water quality measurements and geological logs etc. Analysis of this data is best achieved where a computerized database is used to store and validate the information.

Environmental and groundwater databases vary in sophistication depending on the analysis and reporting that is required for a specific mining operation. In certain instances, the database consists of a spreadsheet, in other instances the database is fully customized with a GIS interface. Although the database functionality may vary between operations and organizations, it is recommended that there are a number of common features as follows:

- The database should have an open architecture to allow data exchange between various software packages particularly where data input into specialist packages are required.
- The software should be user friendly to permit ease of use and reporting. The more difficult the database functionality, the less likely the system will be used by the various stakeholders.
- The data capture portion of the software should contain validation routines to ensure that there is excellent data integrity and confidence in the manipulated information extracted from the database.
- The reporting from the database should be flexible to accommodate changing requirements from management, the authorities and various stakeholders.
- The information contained within the database should be linked to a GIS so that anomalous or problematic areas may be viewed within their spatial context.

- The software platform for the database should be well established to ensure continued support from the suppliers over time.
- The database should ideally include fundamental groundwater analysis tools such as piper diagrams.

It is felt that the investment in collecting the monitoring data will be protected if the historical data is stored in a useful format that will permit a trend analysis to assist for the prediction of future groundwater contamination migration in the immediate vicinity of the site.

B2.3.2 Aquifer Vulnerability and Classification Schemes

Considerable effort has been spent on the development of aquifer vulnerability and aquifer management systems. Three classification systems which are relevant to groundwater site characterization are presented below and while these may be useful within a screening level type of study, they do not follow the risk-based approach of understanding and following the linkage between source, pathway and receptor and should therefore be used with caution.

B2.3.2.1 A South African Aquifer System Management Classification by Parsons (1995).

This document outlines the definition of various aquifer categories that may be used to determine the importance of aquifers for decision making processes. The following table indicates the definitions of Aquifer System Management Classes:

Table B2: Definitions of Aquifer System Management Classes (after Parsons, 1995)

Sole Source Aquifer	An aquifer which is used to supply 50% or more of domestic water for a given area, and for which there are no reasonable available alternative sources should the aquifer be impacted upon or depleted. Aquifer yields and natural water quality are immaterial.
Major Aquifer System	Highly permeable formations, usually with a known or probable presence of significant fracturing. They may be highly productive and able to support large abstractions for public supply and other purposes. Water quality is generally very good (less than 150 mS/m).
Minor Aquifer System	These can be fractured or potentially fractured rocks which do not have a high primary permeability, or other formations of variable permeability. Aquifer extent may be limited and water quality variable. Although these aquifers seldom produce large quantities of water, they are important both local supplies and in supplying base flow for rivers.
Non-Aquifer System	These are the formations with negligible permeability that are generally regarded as not containing groundwater in exploitable quantities. Water quality may also be such that it renders the aquifer as unusable. However, groundwater flow through such rocks, although imperceptible, does take place, and needs to be considered when assessing the risk associated with persistent pollutants.
Special Aquifer System	An aquifer designated as such by the Minister of Water Affairs, after due process.

The reader is referred to Parsons (1995) further documentation on this aquifer classification system.

B2.3.2.2 Waste Aquifer Separation Principle (WASP) by Parsons and Jolly (1994)

Aquifer vulnerability is defined by three factors:

- The nature of the strata.
- Nature of the soil (defined in terms of attenuation potential and permeability).
- Depth to groundwater.

One method of assessing aquifer vulnerability is the **DRASTIC** method. **DRASTIC** is an acronym which covers all the important factors covering aquifer vulnerability:

- D** - Depth to water, which is actually the unsaturated zone.
- R** - Rainfall recharge is the mechanism to transport the contaminant down to the groundwater.
- A** - Travel time in an aquifer depends on the geological material, which directly influences attenuation.
- S** - Soil (top 2 – 3m) is an excellent attenuation zone and also directly controls infiltration.
- T** - Topography effects ponding, run-off, soil development and groundwater gradients.
- I** - Impact on the vadose zone. Geology of the vadose zone controls attenuation potential and flow paths.
- C** - Hydraulic conductivity defines the rates that polluted groundwater can move through the aquifer thus contaminating other areas.

If groundwater exists the risk depends not only on vulnerability, but also on the contaminant load. There are four factors affecting the groundwater pollution load:

- The class of contaminant.
- The intensity of contamination.
- Mode of contaminant disposal.
- Duration of contaminant disposal.

To determine each site's suitability for waste deposition and to rank the sites accordingly, the **Waste Aquifer Separation Principle (WASP)** was used (Parsons and Jolly, 1994). This method was developed through the evaluation of several site selection methods and is to a large extent based on the **DRASTIC** Method.

Three factors are regarded as being important in the assessment of site suitability for waste disposal.

- **The Threat Factor.** All waste disposal sites have the potential to produce leachate at some point in their life and therefore poses a threat to groundwater. The threat is a function of both the volume and the quality of that leachate. A threat factor score is obtained using the designed final area of the site and the type of waste being disposed of.
- **The Barrier Factor.** The unsaturated zone represents the barrier between a waste body and the aquifer. It is within this zone that much attenuation of leachate occurs. Attenuation is extremely complex to model. In this process therefore the time that leachate would take to travel from the base of the waste body to the top of the aquifer, would be used to quantify the ability of the barrier zone to separate the waste from the aquifer. Travel time is calculated using Darcy's law. The data required for this calculation is depth to water, the hydraulic conductivity and the porosity of the vadose zone. The Barrier Factor score is obtained by comparing the calculated travel time to a rating curve.
- **The Resource Factor.** The quantification of the Resource Factor is often the most difficult. The strategic value of an aquifer to a user or potential user is considered. Two sets of questions need to be answered, the first set dealing with current usage and the second with potential usage.

Once scores for all three factors have been determined, the **WASP Index** is computed. It is important to note that the index is used to rank sites within a study area relative to each other rather than determining an absolute value that is transferable from area to area.

B2.3.3 Analytical techniques

There are a number of preliminary analytical techniques that may be used as a preliminary assessment of the hydrogeological conditions underlying the site. These methods may be sub-divided into three categories namely:

- The interpolation of groundwater gradients and groundwater qualities.
- Calculation of the rate of contaminant migration.
- Predicting Groundwater Level Recovery

Groundwater contamination migration from mine workings is usually dependant on the rate of groundwater level recovery. Predicting the rate of groundwater recovery

is therefore also discussed as an important analytical technique for the prediction of groundwater contaminant migration.

The possible analytical techniques that may be used as a preliminary assessment in the mining environment are too numerous to document here. However, it is suggested that preference be given to those analytical procedures that have been published in the scientific literature since they have already been subjected to peer review. Analytical techniques may be used in conjunction with GIS applications to address the spatial variability across the area under investigation. Examples of some of the available techniques that may be used are documented as follows:

B2.3.3.1 Interpolation of Groundwater Gradients and Groundwater Qualities

Groundwater gradients form the foundation for any prediction of the rate and direction of contaminant migration. The appropriate interpolation technique is therefore important for the subsequent management decisions that may be taken. The most fundamental or simplistic method to interpolate groundwater gradients is through the method of triangulation of the groundwater elevations between three monitoring wells. This technique is sufficient to determine the initial groundwater flow direction but may be used as a first estimate only.

A second method of interpolating groundwater gradients involves statistical techniques such as inverse distance and kriging. The later technique is difficult to use for groundwater contaminant investigations since there is usually insufficient information to form an acceptable semi-variogram. For this reason, the majority of the groundwater gradient interpolations within aquifers where the groundwater flow does not necessarily mimic the topography are undertaken with inverse distance. Such interpolated groundwater contours are useful as a first estimate. However it is important to note that such interpolations will not necessarily take any geological structures into account unless the interpolation is restricted to known groundwater compartments.

It is generally accepted that the groundwater gradients mimic the topography in fractured rock aquifers excluding Karstic dolomites. The Institute for Groundwater Studies developed the technique of Bayesian estimation whereby the relationship between the observed groundwater levels and the topographic elevation is used to interpolate the groundwater elevations to topographic contours digitized from existing maps. This technique is useful since it

utilizes the knowledge of the groundwater gradients in the interpolation technique. However, the linear relationship between the topographic elevation and the groundwater gradient should be verified by means of a scatter plot before Bayesian estimation is used. Bayesian Interpolation usually forms the initial water levels for subsequent groundwater modelling studies.

B2.3.3.2 Analytical Calculation of the Groundwater Contaminant Migration Rate

Advective groundwater contaminant migration is based on the following calculation:

$$V_a = \frac{K(h_1 - h_2)/L}{n}$$

where V_a = Actual water Velocity (m/day)
 K = (hydraulic conductivity m/day)
 $(h_1 - h_2)/L$ = Groundwater Gradient
 n = Effective Porosity as a percentage or fraction

This equation may be used to determine a first estimate of the rate of groundwater contaminant migration. However, it must be emphasized that such calculations should not form the basis for remedial measures or other management decisions.

B2.3.3.3 Predicting Groundwater Level Recovery

Until the groundwater gradients have recovered to their ambient groundwater elevations, the direction of groundwater contaminant migration is towards the mine workings. Contaminant migration tends to move away from the mine workings once the surface decant point is reached or the ambient groundwater gradients are re-established. Prediction of the rate of groundwater level recovery is dependant on three factors namely:

- The groundwater ingress into the workings
- The storage capacity of the mine voids and the surrounding host rock
- The permeability of the host rock to permit groundwater ingress into the workings.

Groundwater Ingress into the Mine Workings

It has been assumed that in certain instances the pumping rate from the workings equates to the groundwater ingress. The motivation for this approximation is that

the water level within the workings is relatively constant. However, even small changes in groundwater level represent vast changes in water volume especially where there are extensive interconnections between mines or a number of reefs or seams have been mined. The volume of water that is taken up or released from the mine void storage is therefore critically important.

A simple analytical manner to determine the groundwater ingress into the mine workings adds the volume of water released from or taken into storage in the mine voids to the pumping volume from the workings. The volume of water in storage is determined by calculating the mine void volume with depth and continuously monitoring the water level in the mine workings as part of the ongoing monitoring program.

Other methods of determining the groundwater ingress into the mine workings include:

- Saturated Volume Fluctuation Calculation
- Environmental Isotopes
- Cumulative Rainfall Departures
- Chloride Profiles

A detailed description of these various techniques is beyond the scope of this document. The reader is referred to Bredenkamp et al (1995) for further details on estimating groundwater recharge on the basis of the following techniques. However, it is important to note that the saturated volume fluctuation calculations and cumulative departures are dependant on time-dependant groundwater level, abstraction and rainfall data.

Groundwater ingress may also be determined by estimating recharge volumes to undisturbed strata as opposed to areas of subsidence. Delineating the potential for subsidence in a GIS package is an important consideration for determining the recharge to the underground workings. Mining induced cracks are another set of features that may contribute to groundwater ingress into the mine workings. It is important to identify the volume of surface runoff that may flow into these cracks during various rainfall events.

Storage Capacity of Mine Workings

Storage capacity in the form of mining voids within the mine workings is dependent on the type of the deposit being mined and the mining method. The basic starting point to determine the storage capacity within the mine workings involves a detailed analysis of the mine plans.

The following assumptions may be used for various types of deposits:

- Witwatersrand Gold mines vary from steeply dipping reefs in the Central and Western Basins to more gently dipping reefs in the Orange Free State gold mines. One of the critical issues regarding the dip of the reefs when calculating the mining void volume is the tendency for reef closure for flat reefs located at great depth. Site knowledge is useful in identifying the degree of closure that may be expected. Closure effectively influences the mining height that is used to determine the mine void volume. The mine void volumes are best determined from Digital Terrain Models (DTM's), which take the elevation and true dip of the reef into account. Void volumes are then calculated per 5m intervals so that the variation in void volume with depth may be considered as the groundwater level recovers within the mine workings.
- Coal seams are generally flat in South Africa. Under these circumstances, the void volume is calculated by determining the area of the mine workings and multiplying this figure by the mining height. However, the amount of void space remaining in the mine is dependant on the mining method. In the case of "Bord and Pillar" mining, the void space in the total mine volume will be in the order of 30% to 80%.
- High extraction mining coal mining results in the goafing or collapse of the roof. The original mining void is distributed through the roof. However, the vertical distribution of the final voids is dependant on the mining width of the panels, the depth of mining, the mining height and the competency of the strata. Void spaces above the high extraction panels therefore need to be recalculated as a function of elevation so that this variation may be considered as the groundwater levels recover following mine closure. Hodgson & Krantz (1998) summarize the void volume distribution for high extraction panels as follows. Porosity values for the collapsed material are estimated to represent 30% of the original void space within the coal seam horizon. Where surface subsidence occurs, the drop in the strata is generally 50% of the mining height. The remainder of the original void space, i.e. 20%, is therefore distributed within the zone between the extensively collapsed panel roof and the surface.

Rehabilitated opencast mines usually have a storage capacity in the order of 25%. This figure is used to assign the storage capacity for the pit volumes that are determined from mine plans. However, such estimates

should generally be refined by detailed water balances and flow measurements. Similarly flow measurements or abstraction rates and water levels are important to determine the storage capacity of the mine on a regional scale rather than detailed sampling or analysis on a more detailed scale.

B2.3.4 Graphical techniques

Graphical techniques for groundwater investigations have traditionally included geological surface maps, underground plans and geological cross-sections of the area of interest. However, the greatest requirement for the pictorial representation of geohydrogeological data is for the interpretation of chemical analyses that are generated from the groundwater monitoring network.

Groundwater chemistry is altered as it moves through the hydrological cycle due to processes such as oxidation/reduction, ion exchange and adsorption etc. Apart from the natural evolution of groundwater, chemical changes also occur due to man made activities such as mining. According to the ASTM standard D5877, an understanding of the relationship between the similarities and differences of these water samples is simplified by the use of data analytical methods and the display of these results as pictorial diagrams. However, it is imperative that the chemical analyses are checked for inconsistencies such as ion balance errors before they are included in the dataset.

Lloyd and Heathcote (1985) suggest that the representation of data for interpretation consist of a two-fold approach which should consist of maps indicating the distribution of the parameters and secondly of relevant parameter relationship diagrams. The most frequently used distribution maps are those depicting Total Dissolved Solids or Electrical Conductivity. Such maps may be useful as a cross-check for the interpolation of groundwater flow gradients.

A selection of graphical methods as described in the ASTM Guideline D5877 for displaying results of chemical analyses of groundwater and Lloyd and Heathcote (1985) are as follows:

B2.3.4.1 Box and Whisker Plots

Box and Whisker plots are useful when the attributes of a number of chemical samples are being compared and the median of the sample is shown as a center line within the box. The spread of the sample is shown by the height of the 75th quartile box above and the 25th quartile below

the median line. The skewness of the sample is shown by the relative size of the 75th quartile box in relation to the 25th percentile box. Anomalous or unusual samples are indicated by the length of the whiskers.

B2.3.4.2 Scatter Plots for Correlation Coefficient

This type of scatter plot is a simple X and Y plot of two variables used as a visual tool to depict the correlation between two parameters. The correlation coefficient is a measure of the strength of association or goodness of fit between two continuous variables. The reader is referred to ASTM Guideline D5877 for the calculations of correlation coefficients. Such plots are often used to determine whether there is a linear trend between the topographic elevation and groundwater contours which is a precursor test for Bayesian Interpolation.

B2.3.4.3 Time or Trend Series Plots

The basic time dependant plots consist of the independent variable, time, plotted on the X axis and the dependant variables namely the chemical parameters plotted on the Y axis. Other useful time dependant plots include that of rainfall and water levels for example. Trend lines are often superimposed on these plots to indicate rising or falling trends with respect to groundwater quality.

B2.3.4.4 Schoeller or Vertical Scale Diagrams

According to the ASTM standard D5877, the Schoeller diagram is structured so that the concentrations of the ions are plotted as logarithmic values on the vertical scale. The related horizontal axis of the diagram has no numerical scale and represents the individual ions spaced at even intervals. These ions can be arranged along the horizontal axis in a user-preferred order. This type of diagram allows for the cation and anion constituents of any ground-water analyses to be shown on one diagram so that similarities and differences can be visually evaluated.

B2.3.4.5 Piper Diagrams

In the Piper diagram, the major ions are plotted in the two base triangles of the diagram as cation and anion percentages of milliequivalents per litre. Total cations and anions are considered as 100%. The respective cation and anion locations for an analysis are projected into the rectangle which represents the total ion relationships. Piper diagrams are used to make comparisons between large numbers of samples and may in certain instances be used to calculate the resultant mixing of

low groundwaters if the samples plot on a straight line in each of the three fields. However, a drawback of all trilinear diagrams is that the actual ion concentrations are not shown in the analysis.

B2.3.4.6 Durov Diagrams

The Durov represents an alternative diagram which is normally based upon percentage major ion milliequivalent values but in this case the anions and cations together total 100 percent. The cation and anion values are plotted in the appropriate triangular field and projected into the square main field.

According to Lloyd and Heathcote (1985), the expanded Durov has the advantage over the Piper diagram in that it provides a better display of the hydrochemical types and processes.

B2.3.5 Numerical Modelling

Preliminary calculations for the rate of groundwater contaminant transport have been discussed as a first estimate of the potential impact of mining operations on the surrounding environment. However, such estimates are limited since they cannot accommodate changes in the mining operations and other factors that may influence the geohydrological characteristics of the site. For these reasons, more sophisticated tools are required for the prediction of groundwater contaminant migration and the optimization of future remedial measures. Such tools involve the use of numerical modelling.

According to Anderson and Woessner (1992), modelling serves a dual purpose namely to understand the current geohydrological flow regime and to predict the consequences or a proposed action such as the onset of mining or the implementation of remedial measures. Groundwater contamination models are commonly used to:

- Interpret and Understand Observed Concentrations
- Predict the extent of future groundwater contamination plumes
- Design groundwater monitoring programs
- Assist in the siting of mine residue facilities
- Assist in developing more environmentally responsible mining practices
- Identify and plan potential remedial measures

While groundwater models are based on the groundwater flow equation, each modelling package has certain coding approaches that influence the outcome of the modelling

results. It is important to note that as software developers improve the graphical user interface of groundwater models and the quality of the outputs, more uneducated or inexperienced individuals are misusing models without understanding the inner workings and assumptions on which the modelling code is based. There are a set of modelling protocols as set out in Anderson and Woessner (1992) that are often ignored in the interests of completing the project at hand. However, these modelling protocols and the recommended calibration procedures ensure a degree of quality assurance for the modelling results.

Although there are various groundwater models such as MODFLOW and FEFLOW that are available, the most successful modelling predictions have been based on a sound conceptual groundwater model. The importance of groundwater conceptual modelling has been discussed elsewhere in this document. This forms the foundation for all the subsequent model simulations.

There are a number of groundwater models available as follows:

- Finite element
- Finite Difference
- Two Dimensional
- Pseudo Three Dimensional
- Full Three Dimensional

B2.3.5.1 Comparison Between Finite Difference and Finite Element Models

Finite element models involve the discretization of the model domain into a series of triangles. The frequency and size of the triangles may vary which allows for a greater degree of flexibility in the setup of the model domain for areas where the groundwater gradients may be locally very steep such as in the vicinity of dewatering cones around active mine workings.

Finite difference models discretize the model domain into a series of blocks. Although the block size for a specific column or row may be variable, the size must be consistent across the extent of the model domain. A row may vary in size to other rows within the model domain but the size of the individual row is consistent.

Finite difference models tend to be more popular since they are easier to set up and they are better understood within the international groundwater community with specific reference to MODFLOW. Finite element models offer more flexibility but are harder to populate in terms of information. Alterations to the model setup in finite

element models are also more difficult. The use of finite difference or finite element models may be a project specific consideration and consultancies in the United States tend to use both MODFLOW and FEFLOW depending on the application and the budgetary constraints.

B2.3.5.2 2D Modelling Versus 3D Modelling

The dimension of the modelling domain is dependant on the objective of the modelling exercise. Two-dimensional models are commonly undertaken where a regional flow model or contaminant transport model is undertaken. In this instance, the thickness of the significant aquifers is considerably less than the horizontal dimensions of the area of interest. Such a model may also be considered if there is no significant vertical distinction between the different hydrostratigraphic units within the modelling domain such as a fractured shale and fractured sandstone.

Three-dimensional models are usually undertaken when relatively small scale problems are considered such as the evaluation of remedial measures. This form of modelling will also be important where distinctive hydrostratigraphic units are identified. An example of such a situation would include the near surface weathered aquifer that has lost the majority of its original lithological fabric and the deeper fractured Karoo aquifer where the horizontal permeability is considerably greater than the vertical permeability. Three-dimensional models would certainly be considered for mine residue deposits where the seepage from a potential contamination source is also considered. Most applications involving the simulation of groundwater contamination plumes will ideally be undertaken as a 3D groundwater model. An exception is the contamination flow through a rehabilitated opencast pit where the permeability of the surrounding host rock is so contrastingly low that it has little outcome on the modelling results. Under these circumstances the surrounding undisturbed strata would constitute a no flow boundary.

A special case of two-dimensional modelling is the profile model which represents a vertical cross-section through the model domain. Such models are important where vertical groundwater flow is significant but a full three dimensional model is unwarranted. Another important application of the profile model is the testing of the three dimensional model design. For example, if a dolerite sill has intruded the Karoo strata, the significance of the dolerite sill as a separate hydrostratigraphic unit

will depend on the influence that this intrusion exerts on the localized groundwater contaminant migration patterns. The significance of such a feature may be tested by means of a profile model before a detailed three dimensional model is constructed to consider this feature.

Finite element models are completely 2D or 3D depending on the software that is used. The finite element models tend to be completely 3D in that the model mesh is set up in 3D while finite difference models such as MODFLOW are pseudo-3D in that the hydrostratigraphic units are represented as layers that have a vertical leakage calculated between them.

While finite element models such as FEFLOW are more flexible, they are time consuming to set up and as such may not justify the additional level of detail that may be included in such a model. MODFLOW and MT3D, the groundwater contaminant transport model are well understood and the numerical solutions are proven for applications internationally. Modelling simulations using MODFLOW and MT3D are more rapid without a significant compromise in the resolution of the simulated plumes for the majority of groundwater investigations undertaken in South Africa. However, numerical problems may occur in MODFLOW/ MT3D solutions where very thin aquitards or aquicludes or geological structures are considered. The use of either modelling approach will depend on the application and the detail of field information that is available.

B2.3.5.3 Definition of the Modelling Domain

According to Anderson and Woessner (1992), numerical groundwater models consist of the governing groundwater equation, model boundary conditions and initial conditions. The groundwater modelling domain is defined by aquifer boundaries which should ideally coincide with natural geohydrological boundaries. Such boundaries include:

- Type 1: Specified Head Boundaries (Dirichlet conditions) a groundwater elevation or head is fixed under these conditions such as a large lake or dam.
- Type 2: Specified Flow Boundaries (Neumann conditions) for which a flux is given. A no-flow boundary or watershed is a specific condition of this type since the flux is set to zero.
- Type 3: Head-Dependant Flow Boundaries (Cauchy or Mixed Boundary Conditions) In this instance the flux across the boundary is dependant on the driving groundwater head or elevation.

Most groundwater modelling software packages can accommodate these boundary conditions. However, it is the incorrect definition of the model domain boundaries that results in groundwater simulation errors.

Internal boundaries such as river systems have been addressed in the past by assigning constant head or groundwater elevations at these points in the model domain. However, this situation does not allow for variations in the groundwater elevation during summer and winter months. For this reason, more recent software developments cater for the more accurate representation of the surface water systems within the model domain.

B2.3.5.4 Assigning the Model Aquifer Parameters

All groundwater contaminant transport models consist of two components, namely the groundwater flow component and the groundwater contaminant migration component. The groundwater flow velocities generated by the groundwater flow model are input into the subsequent groundwater transport models.

The model aquifer parameters therefore consider both the input parameters for the groundwater flow and groundwater transport modelling. A synopsis of the groundwater model input parameters are indicated in Table B3.

The major factor that influences advective contaminant transport is the permeability and the groundwater gradients. Although there is greater uncertainty associated with aquifer parameters such as dispersion, this parameter has a lesser influence on the simulated contamination plume extent in the majority of cases. However, dispersion will influence the simulated concentrations at the leading edge of the plume.

Observed or estimated aquifer parameters should be interpolated within the specific hydrostratigraphic units as defined by the groundwater conceptual model. Tendencies to interpolate aquifer parameters across such units may lead to groundwater model oscillations at best and inaccurate groundwater contamination migration simulations at worst. The interpolation of the input aquifer parameters and groundwater gradients is discussed elsewhere in this document.

Uncertainties in the modelling input parameters are addressed by means of Monte Carlo simulations. Random values are generated from population distributions for the modelling input parameters to be used in subsequent

simulations. The simulated groundwater contamination plumes are analyzed as a probability density function to determine the pollution risk for an exceedance probability percentage of 5% (95% confidence interval), 20% (80% confidence interval) and 40% (60% confidence interval) for critical points of concern such as the nearest receptor.

Table B3: Origin of Groundwater Modelling Input Parameters

Parameter	Source	Comments
Horizontal Permeability	Measured	Determined from Pumping Tests and Numerical Models as discussed in Chiang and Riemann (2001)
Vertical Permeability	Estimated Measured Simulated	This parameter will be initially estimated. Where observation wells are available this parameter may be determined from pump testing. The final value for this parameter is usually simulated. These techniques are discussed in Chiang and Riemann (2001).
Storage Coefficient	Estimated Measured	This parameter is initially estimated from previous investigations. Storage Coefficient may be measured by means of pumping tests. However, fractured aquifer characteristics must be considered as discussed in van Tonder and Bardenhagen (2001).
Rainfall Recharge	Estimated Calculated	Estimated from previous investigations or calculated from field observations
Longitudinal Dispersion	Estimated Measured	This parameter may be determined from tracer tests. However, it may be estimated to be one tenth of the total anticipated plume migration where there are budgetary constraints.
Transverse Dispersion	Estimated	This parameter may be determined from tracer tests. However, the parameter is estimated to be one tenth of the longitudinal dispersion on projects where there are budgetary constraints.

B2.3.5.5 Assigning the Contamination Source Concentrations

The contaminant source concentrations from mining operations and the associated infrastructure has traditionally been taken to be the highest observed groundwater concentrations in the monitoring borehole drilled adjacent to the source or the current groundwater concentrations in the immediate vicinity if time dependant data is not available. The problem with such an assumption is that the source concentration is assumed to be relatively constant over time. However, it is known that this is not the case for pollution sources still undergoing active sulphide oxidation. The simulated plume concentrations may therefore overestimate or underestimate the impact on the groundwater due to an incorrect estimate of the source concentration over time.

It is recommended that the geochemical predictions of the source concentrations be included in the groundwater simulations as a multi-species variable source for the duration of the proposed model simulations. This would allow for a more realistic quantitative simulation of the future groundwater quality than is currently being undertaken.

B2.3.5.6 Assigning Plume Retardation Factors

Geochemical models have been used to determine the variation in the source concentrations from potential mining-related contamination sources. This variation in chemistry may be input into the groundwater modelling packages such as MT3D for multi species. However, there are essentially six types of reactions that may retard or alter the groundwater contaminant plume as it migrates through the subsurface. These are as follows:

- Adsorption-desorption reactions
- Acid-base reactions
- Solution-precipitation reactions
- Oxidation-reduction reactions
- Ion-pairing or complexation (Aqueous Speciation)
- Microbial synthesis

It is possible that the influence of these reactions on the groundwater contaminants may be determined by means of geochemical modelling. However, groundwater modelling packages commonly only have a retardation factor included in the contaminant transport modelling. The obvious problem with such a simplistic approach is

that it will not take the subsequent solution of secondary minerals into account.

Groundwater transport models are therefore used as a tool to optimize remedial measures based on conservative or worst case scenario groundwater contamination migration simulations. The impact of mining related contamination may be over stated although the sulphate plume, which is the dominant ion, is only retarded by some 25%.

A manner in which the prediction of groundwater contamination migration may be improved upon would be the determination of retardation factors for various chemical constituents across the model domain. These retardation factors would be global parameters that would not necessarily account for the chemical interaction between the various species within the groundwater contaminant plume.

B2.3.5.7 Groundwater Model Calibration

Contaminant transport simulations utilize the groundwater flow velocities and directions as key model inputs. These values are combined with the contaminant transport parameters such as longitudinal and transverse dispersion to produce the resultant rate of contaminant migration.

Given the relative complexity of the model simulations, it is imperative that the simulation results reasonably reflect the physical observations. This is determined by means of model calibration which demonstrates that there is a reasonable correlation between:

- The simulated and observed groundwater elevations at specific monitoring boreholes.
- The simulated and interpolated groundwater gradients to ensure the correct shape of the groundwater flow regime.
- The simulated and observed groundwater chemistry for conservative or non-retarded chemical constituents.

Without model calibration, the groundwater model simulation results may only be viewed as qualitative results rather than quantitative concentrations based on a variable pollution source.

B2.3.5.8 Recommended Future Groundwater Modelling Approaches

Numerical modelling is the most effective tool in predicting the groundwater contamination migration and its impact on the environment. It also represents one of the most useful tools in evaluating a study area or remediation

measures before expensive field investigations and or pilot studies are undertaken. However, a varying source term as predicted from geochemical principles needs to be included in the definition of the source concentration. It must also be borne in mind that the simulated concentrations of the groundwater plume are also conservative unless overall retardation factors may be determined from geochemical modelling for the study area. Reactive transport models (e.g. PHT3D, PHAST and the React Modules in GWB Pro) are available to properly assess the fate of contaminants originating from a source term as it moves along its transport pathway to the critical receptor.

B3 GEOCHEMICAL TOOLS

For the purpose of this BPG, the geochemical tools are divided into the following two categories:

- Geochemical analytical techniques
- Geochemical models

The reader's attention is drawn to the distinction made between the role of the analytical techniques and the models in the introduction to Chapter 6 of this BPG. An understanding of the respective roles of the different techniques within the assessment process will show that, in most instances, it will be necessary to use a combination of the available geochemical tools.

B3.1 GEOCHEMICAL ANALYTICAL TECHNIQUES

Geochemical analytical techniques are an essential and integral part of all geochemical assessments. Different analytical techniques have been developed to provide different types of data. As there is no single analytical technique that is capable of assessing all the critical chemical, physical and microbiological factors that affect sulphide oxidation and neutralisation of ARD, **any credible geochemical assessment will necessarily comprise of a range of analytical techniques.**

It has traditionally been common practice to divide geochemical analytical techniques into two major categories, viz. static and kinetic tests, and this distinction will be maintained in this BPG. In addition, there are other physical and mineralogical analytical techniques that are also an essential component of all geochemical assessments.

The analytical techniques that are covered in this section of the BPG are the following:

- Mineralogical analyses
- Major and Trace Element Analysis
- Particle Size Distribution
- Acid Base Accounting (acid potential, neutralisation potential, interpretation of ABA)
- Kinetic Tests (Humidity Cells, Column Leach Tests, Kinetic Simulations)

This BPG presents a short summary discussion of the different analytical techniques and does not go into detail. The interested reader can obtain a very comprehensive review of all the available techniques including their limitations, advantages and role in ARD prediction in the Handbook for Waste Rock Sampling Techniques, SENES (1994). Additional references are also given in the text below.

B3.1.1 Mineralogy

A review of the basic chemical equations involved in sulphide mineral oxidation and dissolution of neutralising minerals (see Chapter 3 of this BPG) clearly indicates that mineralogy is the key factor that needs to be understood before any predictions of future water quality can be made. Despite the clear evidence for the importance of mineralogy, it is often wrongly ignored in favour of more simplistic analytical techniques such as ABA. This BPG aims to rectify this situation by clearly emphasizing the essential role that mineralogy must play in geochemical assessments. A summary of petrographic and mineralogical assessment techniques is given by Shaw & Mills (1998) while additional detail is provided by Jambor & Blowes (1998).

The mineralogical analyses are structured to provide the following types of data:

- The type and distribution of minerals in the material being assessed
- Grain size distributions
- The mineral forms that occur (e.g. massive, nodular, disseminated, etc.)

According to Downing & Mills (1998), petrographic studies should be conducted as part of the ABA procedure with the following general objectives:

- To determine primary and secondary mineralogy and alteration variation that would impact the determination of neutralisation potential.

- To examine sulphide mineralogy.
- To determine modal mineralogy for rock classification as a check on field classification used in drill logs.
- To examine grain size and grain boundaries as reaction sites for acid rock generation.

Petrographic or mineralogical examination of samples is usually conducted by transmitted and reflected light microscopy and by various X-ray diffraction (XRD) techniques. Additional techniques that can be employed to answer specific questions are electron probe microanalysis (EPMA) and scanning electron microscopy (SEM).

The data obtained from a mineralogical assessment programme will enable the interpretation of kinetic test data and will also give essential guidance as to which primary and secondary minerals play a controlling role in the water chemistry - this is essential knowledge in all geochemical modelling exercises.

B3.1.2 Major & trace element analysis

A knowledge of the elemental content of the material being assessed is essential. This analysis can be undertaken using various analytical techniques although the most common methods would generally be acid digestion followed by multi-element determination by ICP/mass spectrometry or X-ray fluorescence (XRF) spectrometry. Additional analyses may be required to obtain a proper assessment of the sulphur species as discussed in Section B3.1.4 below.

In addition, it is normally necessary to determine the nature and quantities of soluble constituents that may be washed from materials under natural rainfall conditions and a range of test procedures are available to determine this. A summary of these test procedures is given by Mills (1998d).

Knowledge of trace elements is important for different reasons including the following:

- "The incorporation of both interstitial impurities and mineral inclusions in pyrite will cause local strain in the crystal structure, rendering the pyrite more susceptible to alteration", (Kwong, 1993).
- Trace elements in mine waters may be toxic and very difficult and expensive to remove - strategies to prevent liberation of these trace elements are preferable.

B3.1.3 Particle size distribution

Knowledge of particle size distributions of both the sulphide and neutralising minerals is essential for any geochemical assessment or prediction exercise. As the sulphide oxidation and ARD neutralisation reactions occur at the material surface, knowledge of the surface area is essential and the surface area is calculated from particle size analyses. As discussed in Appendix C of this BPG, most kinetic geochemical models will also distinguish between coarse and fine particles in terms of how they are modelled (fine particles are normally considered to be those < 2 mm).

Most mining sites that require geochemical assessment will exhibit a very wide range of particle sizes from sub-micron size to large boulders. The exception is tailings dams or slurry ponds where the particle size distribution is narrow and practically all the particle can be considered to be fine. A discussion of particle size issues is given in Mills (1999b). When undertaking predictive assessments, it is also necessary to consider the fact that particle size distributions may change with time as smaller particles are washed out or consumed and larger particles are physically and chemically weathered to again produce smaller particles. Weathering effects may be different for different materials, e.g. pyrite may be more friable and weather more rapidly than carbonate minerals.

There are varying analytical techniques for determining particle size that range from physical screening processes for the larger particles to sophisticated techniques for the smaller and sub-micron samples.

An additional particle size related feature that is very important in terms of pollution prediction exercises is that of mineral liberation size (Mills, 1999b). This is the size to which an ore must be crushed or ground to produce separate particles of either value mineral or gangue that can be removed from the ore. The relevance of this with regard to pollution prediction assessments is that reactivity of the mineral will increase as the liberation size approaches the grain size of the pyrite particle.

It is also very important to understand the effect of analytical techniques such as ABA and humidity cells where the naturally occurring particle size is dramatically altered by grinding of the sample to a very small size that will increase the effective surface area and reactivity of the material by orders of magnitude. It is precisely this dramatic effect on particle size that severely limits the practical predictive value of such analytical techniques. **It is only at 100% liberation that acid generation**

potential as determined by ABA methods equals the acid generation potential of kinetic test or field samples. This is also one of the primary reasons why a range of samples that are all assessed in ABA testing to have the same acid potential could all exhibit widely different behaviour in the field.

Information on liberation size is obtained from quantitative mineralogical and petrographic studies as discussed in Section B3.1.1 above. Mills (1999a) recommends that: *“ARD programs should include liberation studies in all but the most straightforward cases, and particularly in cases where predictions from static and kinetic testwork are inconsistent or contradictory”.*

B3.1.4 Acid base accounting (ABA)

Acid Base Accounting (ABA) has traditionally been the mainstay of many geochemical assessments and many books have been written and debates have been held about the different ABA techniques and their application and limitations. Although ABA should be considered as a very valuable and essential component of most geochemical assessments, it has unfortunately also been an analytical test that has been grossly misapplied and credited with predictive powers that it clearly does not have. It is opportune to reiterate the points made in Chapter 3 of this BPG and to emphasize that **the geochemical behaviour of a material is influenced by a multitude of factors that are not at all considered or evaluated within an ABA test.** These factors that are ignored by the ABA test include:

- climate (rainfall, temperature, evaporation)
- surface area (particle size)
- mineral liberation size
- porosity of material
- hydrology (infiltration)
- geohydrology
- mineralogy (mineral composition, trace elements)
- microbiology
- biological and chemical rates
- oxygen availability

This section of the BPG does not aim to reproduce all the discussion on ABA techniques and will only provide a summary of the key features that are considered important with regard to the objectives of this BPG. A good, well-referenced summary of the available ABA techniques is given by Mills (1999a) with a further well-referenced document on the interpretation of the ABA data also given by Mills (1998a).

ABA is a simple analytical procedure that measures the acid generation potential (AP) and the neutralisation potential (NP) of the sample and then either subtracts the AP from the NP to give a net neutralising potential (NNP) or calculates the NP/AP ratio. These manipulated values are then typically compared with a predetermined or set value in order to distinguish between those samples that are likely to be acid generating those that are not and those indeterminate samples that require further evaluation. As the ABA technique provides no information on the speed (or kinetic rate) with which acid generation or neutralisation will proceed it is referred to as a static test.

The words “acid potential (AP)” and “neutralisation potential (NP)” are used as the ABA technique cannot assess the actual field behaviour of material but can only estimate its potential behaviour, even when combined with detailed mineralogical analyses. The ABA techniques generally determine a “worst case” for the AP and a “worst”, “most likely” or “best” case for NP depending on the method used. the question as to whether these potentials are ever realised requires consideration of all the other factors listed above that influence the geochemical behaviour of a material but that are not considered in the ABA test. The **question as to whether the potential is realised or not, is the essence of the geochemical prediction exercise** and prediction therefore falls outside the capability of the ABA test.

B3.1.4.1 Neutralisation Potential Procedures

Although a variety of simple chemical procedures have been developed to assess the neutralisation potential of a material, this information should always be considered in conjunction with information obtained from detailed mineralogical studies. The need for this is dictated by the widely varying reactivity of the different neutralising minerals and the fact that different NP techniques result in the dissolution of different neutralising minerals. The most common NP determination methods are:

- Sobek Neutralisation Potential Method
- Peroxide Siderite Correction for Sobek (Skousen et al)
- BC Research Inc. Initial Test Procedure
- Lapakko Neutralisation Potential Test Procedure
- Modified Acid Base Accounting Procedure for Neutralisation Potential (Lawrence)

The procedures listed above differ primarily in the degree of aggressiveness of acid reaction with the sample - i.e. the variety of minerals attacked by the acid and the degree of mineral dissolution. The following Table B4 has been prepared by Mills (1998a) to summarise the different procedures and suggests, qualitatively, which minerals are dissolved by each procedure. Detailed references for the data presented in the Table are given by Mills.

Table B4: Procedure Conditions for Neutralisation Potential Determination

PROCEDURE	ACID	AMOUNT OF ACID ADDED	END pH OF ACID ADDITION	TEST DURATION	TEST TEMP.	MINERALS DISSOLVED
Lapakko	Sulphuric	To reach pH 6.0	6.0	Up to 1 week	Ambient	Ca + Mg carbonates
BCRI Initial	Sulphuric	To reach pH 3.5	3.5	16-24h	Ambient	Ca + Mg carbonates Possibly chlorite, limonite
Modified Sobek	Hydrochloric	Determined by Fizz Test	2.0 - 2.5	24h	Ambient	Ca + Mg carbonates Some Fe carbonate, biotite, chlorite, amphibole olivine (forsterite-fayalite)
Sobek	Hydrochloric	Determined by Fizz Test	2.0 - 2.5	Until gas evolution ceases (test including titration up to 3h)	Elevated (c. 90 °C)	Mineral carbonates Ca-feldspar, pyroxene, olivine (forsterite-fayalite) Some feldspars (anorthoclase > orthoclase > albite); ferromagnesians - pyroxene, hornblende, augite, biotite
Sobek - Siderite Correction	Procedure as for Sobek, but with peroxide correction for siderite					Ca + Mg carbonates, excludes Fe + Mn carbonates. Otherwise as per Sobek
Net Carbonate Value (NCV)	Method uses combustion-infrared analysis, not acid digestion					Calcite, dolomite, ankerite, siderite
Inorganic Carbon-Carbonate	Method uses Leco furnace or equivalent, not acid digestion					Mineral carbonates

It should be clear from the information presented in Table B4 that no single NP determination can accurately reflect the true NP in the field and that a mineralogical analysis must be undertaken to enable the proper selection and interpretation of NP results. The MEND Report 1.16.3 states that "In all cases, mineralogy is the key parameter which must be evaluated and considered in a waste characterisation programme. Mills (1998a) proposes that for samples containing carbonates and reactive siderites:

- 1) The Lapakko and Inorganic Carbon-Carbonate methods will tend to give a "worst case" neutralisation potential since the carbonates are credited, but other minerals are not.
- 2) The Sobek method will tend to give a "best case" neutralising potential since all carbonates and other minerals soluble at the lowest pH of the test will be credited.
- 3) The BCRI Initial and modified Sobek methods will tend to give a "most likely case" neutralisation potential since the carbonates and only the most reactive siderites are credited.

Mills cautions however, that even the Lapakko and Inorganic Carbon-Carbonate may overestimate the real or field NP. This is possible because some of the neutralising minerals present may be inaccessible to the ARD because of physical placement, particle size, or because of "armouring" by metal precipitates. It is also important to point out that the Sobek method is considered highly unreproducible due to the subjectivity of the Fizz Test that is associated with the method.

B3.1.4.2 Acid Generation Potential Procedures

The determination of acid potential (AP) is less problematic than the NP methods although there are different types of sulphur minerals that need to be distinguished. Typically, samples are considered to have the potential to include both sulphide minerals and sulphate minerals. Sulphate minerals have no potential to oxidise to sulphuric acid they do not contribute to the AP, although certain sulphate minerals such as melanterite may dissolve, hydrolyse and then generate acidity.

Although techniques are available to determine a wide range of sulphur species, it is typically only necessary to

measure total sulphur, sulphate sulphur and to calculate sulphide sulphur. The AP is determined from the sulphide sulphur. In certain cases where barium is present, it may be necessary to correct for barite.

Calculation of the acid potential through simple measurement of the sulphide sulphur content is, however, an oversimplification with the following primary inherent errors (Pulles, 1998):

- Although some common sulphide minerals such as sphalerite (zns) and galena (pbs) do not produce acid (although they do oxidise to liberate metals) this test will include them as having an acid potential.
- The mineralogy and microscopic structure of the sulphide mineral which is of critical importance in determining the reactivity of the sulphide is ignored.
- The critically important catalytic effect of bacteria is ignored and only chemical oxidation is considered.
- The reaction rates and kinetics of sulphide oxidation and dissolution of base minerals are ignored and the test is undertaken under conditions where complete and rapid reactions occur in a closed system (laboratory beaker) as opposed to the open systems encountered in the field.

B3.1.4.3 Evaluation & Assessment of ABA Data

There are two primary methods of combining the measured NP and AP data to obtain calculated values that can then be evaluated against set criteria:

- Net Neutralising Potential ($NNP = NP - AP$)
- Neutralisation Potential Ratio ($NPR = NP/AP$)

Additional factors that can be considered when evaluating ABA data are the paste pH and the % total sulphur.

The discussion presented in Sections B3.1.4.1 and B3.1.4.2 above clearly indicate that the NNP and NPR values will be directly affected by the analytical method that has been used, particularly for the determination of NP. North American practice and experience has led to the establishment of criteria for NNP and NPR against which the results from a particular material being assessed can be compared in order to determine whether there is a risk of the material generating acid under field conditions. The severe limitations of the ABA test procedures insofar as they ignore most of the fundamental parameters that influence geochemistry, have been previously explained. **Compliance with the NNP and NPR criteria is therefore no guarantee that the material will behave as expected.** Of particular

importance to South African mines is the fact that the NNP and NPR criteria are based on North American empirical field observations that **may not be valid for the typical South African field conditions**, especially as a result of significant climatic differences. The only way in which comparison with preset criteria could confidently be used as a screening assessment in South Africa is if those criteria have been derived from field observations collated from South African situations or North American situations that are demonstrably similar to those encountered in South Africa.

In British Columbia, Canada, the Ministry of Energy and Mines (BC MEM), have defined guidelines for interpretation of NP and AP data as shown in Table B5 below (Price, 1999). However, Price also states that *“Unlike practitioners who rely solely on static test results, BC MEM also requires a knowledge of the type, concentration and reactivity of minerals with neutralisation potential, likely exposure, physical and hydrological conditions, the rate of acid generation, along with kinetic data”*.

Table B5: Neutralisation Potential Ratio (NPR) Screening Criteria

POTENTIAL FOR ARD	INITIAL NPR SCREENING CRITERIA	COMMENTS
Likely	<1:1	Likely ARD generating
Possibly	1:1 - 2:1	Possible ARD generating if NP is insufficiently reactive or is depleted at a faster rate than sulphides
Low	2:1 - 4:1	Not potentially ARD generating unless significant preferential exposure of sulphides along fracture planes, or extremely reactive sulphides in combination with insufficiently reactive NP
None	>4:1	No further ARD testing required unless materials are to be used as a source of alkalinity

In addition, it must be remembered that evaluation of ABA data gives no indication of the dissolved contaminants such as sulphate, sodium, etc. Material that will forever produce alkaline water may still produce high levels of sulphate and certain metals such as manganese. **Where an ABA assessment indicates that future acidity is likely, it cannot predict when the onset of acidity will occur, how acid the water will become and at what concentrations the contaminants such as sulphate and metals will occur.**

For the above reasons, ABA accounting data should be treated with great circumspection and they should never be used on their own to make a prediction of the long term potential for acid production and the long term prediction of water quality. At best, and provided there is a large database of ABA data and that the other important issues discussed above are known and considered, ABA data can be used to give a very rough screening assessment of the potential of a particular sample to produce acid. However, this data will have fairly large error margins and will generally tend to overestimate the neutralisation potential. This approach is not considered suitable for making assessments of the long term risks of water quality problems from practical mining situations and it should most certainly not be considered as presenting a "worst case" scenario.

B3.1.5 Kinetic Tests

There are a range of well-known kinetic tests that are routinely undertaken as part of a geochemical assessment at a mining site (Mills, 1998b):

- Humidity cells
- Column leach tests

In addition a new kinetic test that holds promise - the Oxygen Consumption Test is also briefly described here.

The difference between static and kinetic testwork is that the kinetic tests determine the behaviour of the material over time, thereby generating data that can be presented as a rate function per unit time. All the different kinetic test procedures contain the following elements:

- Subjection of sample to periodic leaching
- Collection of drainage for analysis
- Calculation of rates of acid generation and neutralisation capability depletion
- Calculation of rates of metal release
- Prediction of water quality

The kinetic tests attempt to simulate field conditions in the laboratory. However, this simulation is never very accurate as there are a multitude of field conditions and variability that cannot be simulated in the laboratory. These tests also generally require some crushing of the sample to a smaller particle size in order to accommodate the dimensions of the test apparatus. The key deficiency with kinetic laboratory tests is that, like ABA tests, they have no ability to be confidently used in a predictive sense. The onset of steady state conditions in a laboratory test provides no indication as to whether those steady state conditions might not change a few months or years later with different meteorological conditions or the depletion of certain base minerals.

The kinetic tests become particularly useful if they are used to provide key data for use in subsequent kinetic modelling exercises. Examples of such data are oxidation rates for different size fractions, initial leachate composition, armouring effects, types of secondary minerals formed, etc. In some cases, specially designed kinetic laboratory tests can also provide key data for use in empirically based models.

B3.1.5.1 Humidity Cells

While there are a range of humidity cell designs in use, there is generally a move to standardise with the ASTM method - *Designation: D5744-96 Standard Test Method for Accelerated Weathering of Solid Materials Using a Modified Humidity Cell*. The test procedure involves placing a known mass of material with a defined particle size distribution into a container that is subjected to weekly cycles of three days humid air followed by three days dry air, finally followed by washing with water for a day. The procedure aims to accelerate the natural weathering rates by at least an order of magnitude over field rates. The ASTM Method clearly states, however, that the procedure is not intended to simulate site specific leaching conditions.

Prior to starting a humidity cell test the sample must be screened (generally to below 2mm) and weighed and a particle size analysis should be undertaken in order that sample surface area can be determined. The sample should also be subjected to ABA analysis, mineralogical assessment, ICP metals analysis and major component whole rock analysis. These tests should be repeated on the residue left after completion of the humidity cell test using identical analytical procedures.

Where it is desired to undertake humidity cell testwork on previously oxidised material, special procedures may be required to characterise and remove the previous oxidation products from the sample. A special interpretation is also required in cases where the determination of acid potential has indicated the presence of significant amounts of naturally occurring sulphate-sulphur within the sample as the release of such sulphate will be due to dissolution and not sulphide oxidation.

While the ASTM Method recommends a minimum test duration of 20 weeks, Price (1997) recommends 40 weeks. More recently, Morin and Hutt (1999) summarised the requirement for humidity cell tests as follows:

- 1) 50% of cells can be terminated after 1 year, while the other 50% should be operated for more than 3-5 years to obtain long-term stable rates.
- 2) At least 12 cells are needed to obtain a reasonable indication of average rates, biased towards the lowest measured values. However, 40 or more cells may be needed to obtain a reasonable distribution pattern of values such as normal or log-normal distributions

A humidity cell can provide an indication as to whether a sample will go acid but not when the onset of acidity will occur. It does not provide data that can be used to predict long-term water quality, as the test conditions do not simulate field conditions. Examples of case study interpretation of humidity cell test programmes are given by Mills (1998c).

B3.1.5.2 Column Leach Tests

Leach columns have been developed to provide information on the kinetic behaviour of material in either sub-aerial or sub-aqueous environments. Similarly to humidity cells, the objective is to obtain weekly or monthly leachate samples for quality analyses, although without forced humid/dry air cycles. There is no standard column leach test procedure and these systems are generally set up in an attempt to simulate actual site specific situations. Advantages of these leach columns over the humidity cells are that they can take considerably greater masses of sample and can handle material of a greater particle size. They can also be operated to more closely simulate field climatic conditions. Similar characterisation of the sample before and after the testwork should be undertaken as has been recommended for the humidity cell tests.

B3.1.5.3 Oxygen Consumption Method

A technique, known as the oxygen consumption method (OCM) to determine oxidation rates has recently been proposed by Andersen, et al (1999). OCM involves placing a reactive sulphide rock sample in a sealed chamber and measuring oxygen content of the gas phase in the head space above the sample over a period of 1 to 3 days. This enables rate of oxygen depletion per unit mass of material to be calculated. Effects of particle size, sulphur content, temperature and inoculation with *T ferrooxidans* on rate of oxygen consumption was investigated. The method is reported as being fast and inexpensive compared to humidity cells. Although the method is still new and will require validation before it becomes accepted, it promises a number of advantages over the time-consuming and expensive humidity cell tests.

B3.2 GEOCHEMICAL MODELS

The classification and evaluation of geochemical models for application to acid rock drainage (ARD) is based on other model reviews and compilations that have appeared in the literature. Early reviews include those by Nordstrom et al. (1979) on aqueous equilibrium models

and by EPRI (1984) on solute transport models. Other reviews were done by Grove and Stollenwerk (1987) and Engesgaard and Christensen (1988). Yeh and Triparthi (1989) performed a critical review of the approaches used for transport modelling, and Mangold and Tsang (1991) thoroughly listed and reviewed groundwater flow and transport models. More recent reviews are given by Parkhurst and Plummer (1993) and Alpers and Nordstrom (1995), and contained in Loeppert et al. (1995) and Lichtner et al. (1996). The focus of this review is to summarize the classes of geochemical models and their applicability to prediction of acidic drainage from mine waste rock. It is largely based on the work of Perkins et al. (1995). The review is focused on geochemical components, it covers a broad spectrum of model categories, and it concentrates on a more specific prediction problem, namely acidic waste rock drainage.

The main reasons typically put forward for undertaking predictive geochemical modelling include the following:

- Characterisation and interpretation of current contaminant loads
- Identification of soluble and mobile metals
- Prediction of future contaminant loads and concentrations
- Assessment of future water treatment needs
- Comparison of management and decommissioning options

The science of geochemical modelling is a very complex one for the following reasons:

- 1) Geochemical modelling requires the integration of flow, heat, oxygen and geochemical models.
- 2) Geochemical models (particularly the kinetic models) are very complex and require large amounts of data in order to run. The outputs from geochemical models require expert interpretation.
- 3) There are a number of different models available, each of which is suitable for different applications - the specialist needs to be able to choose the right model and must be capable of applying a range of models.
- 4) Data inputs for geochemical models often have major uncertainties that need to be understood and catered for.

The discussion presented in this BPG aims to summarise the key considerations with regard to geochemical modelling in order to enable the user to identify the

key capabilities and limitations of the different types of models. No specific models are recommended although clear guidance is given on the types of models to be used for different situations. Readers that wish to obtain a very detailed discussion of modelling options are referred to Perkins et al (1995): *Critical Review of Geochemical Processes and Geochemical Models Adaptable for Prediction of Acidic Drainage from Waste Rock*.

B3.2.1 Processes to consider in geochemical modelling

The basic physical, chemical and microbiological processes that govern sulphide oxidation and neutralisation of ARD have been summarised in Appendix A. From the perspective of undertaking predictive geochemical modelling, the following important processes need to be considered and incorporated:

- Acid generation due to biological and chemical oxidation of sulphide minerals.
- Diffusion of oxygen into waste residue deposits and also into the rock particles.
- Convective transport of oxygen into waste residue deposits.
- Consumption of oxygen by carbon oxidation in coal wastes and by sulphide oxidation in all mining scenarios.
- Production and transport of heat (conduction and advection).
- Variation of temperature with depth and effect of temperature on oxidation rates
- Inventory of particle sizes and changes over time due to consumption, washout and weathering
- Release (leaching) of metals and radionuclides.
- Transport of dissolved chemical species (e.g. sulphate, iron, aluminium, calcium, magnesium, radionuclides, etc.)
- Dissolution of solid neutralising minerals (e.g. calcite, dolomite, sericite)
- Formation (precipitation) of secondary minerals (e.g. ferric hydroxide, jarosite)
- Solubility of solid phases (e.g. precipitates, minerals)
- Speciation of dissolved constituents
- Estimation of solution pH
- Solids solution equilibria for metals and radionuclides
- Adsorption of metals and radionuclides onto aluminium and ferric hydroxides, jarosite and organic carbon surfaces

- Coprecipitation (e.g. radium with gypsum, lead-210 with lead sulphate, lead carbonate, etc.)

The first requirement for predicting the composition of acid rock drainage is that geochemical models must accurately represent the important geochemical processes that lead to acid generation. Geochemical processes in waste rock dumps create acidity and release metals by a complex path that commences with the oxidation of sulphide minerals. Although this path is similar to the processes occurring in the unsaturated zone of tailings, the processes occurring within waste rock dumps are driven primarily by the ready exchange of oxygen with the atmosphere and by wetting and drying cycles.

Waste rock dumps are wetted intermittently by infiltration of rainwater and by seasonal runoff; drainage and evaporation dry them. During drying, the relative importance of drainage compared to evaporation is determined by the physical properties of the waste rock dump; these include hydraulic conductivity, suction potential, and the chemical potential gradient between water in the pores and water in the atmosphere. The time to complete the entire wetting-drying cycle is dependent upon porosity, permeability and other physical and climatic factors.

For a waste rock dump, in a region of moderate to high rainfall, a complete cycle comprises four sequential stages:

- (1) An air-oxidative stage that results in the destruction of sulphides and the formation of metal oxyhydroxides and sulphosalts. The nature and extent of the reactions are determined partly by the relative humidity of the air (steger, 1982);
- (2) During water infiltration, some of the pores are wetted by the aqueous phase to the extent that dissolution of primary and secondary minerals occurs. Oxidation of sulphides or sulphosalts proceeds further if the water contains dissolved oxygen;
- (3) During water drainage, solutes dissolved in the pore water are removed from the pore spaces and transported to the water table, or are collected in drainage ditches and in other surface water bodies. Air replaces pore water during drainage and an aqueous-phase film is left behind coating the grains;

- (4) Subsequent evaporation of the water film during the drying cycle results in the precipitation of solids, including additional iron sulphate salts and hydrous secondary iron minerals that coat the sulphide mineral surfaces. As drying continues these minerals may dehydrate, crack and spall from the sulphide surfaces, exposing the sulphides to atmospheric gasses (pratt et al., 1994). However, in an arid climate, the flow of liquid water through a waste rock dump is greatly diminished, and in the case of zero water flow only stage 1 and perhaps stage 4 are important.

The major geochemical processes in waste rock dumps are oxidation, dissolution and precipitation. Four classes of mineral-fluid reaction contribute to acidity and to metal release and consumption:

- (1) Oxidation of sulphides, releasing acid, major and trace metals and sulphate;
- (2) Precipitation of oxyhydroxides, releasing acid and consuming the more insoluble major and trace metals;
- (3) Dissolution/precipitation of sulphate minerals mediating the dissolved metal concentrations as well as tds; and
- (4) Dissolution of oxyhydroxides, carbonates and silicates, thereby consuming acid. Co-precipitation may also provide a major control on trace element concentrations. Dissolution and precipitation are much more important controls on drainage water quality than are ion exchange or adsorption/desorption processes. The key geochemical processes at work in sulphide oxidation and acid neutralisation situations can be summarised as shown in table b6.

Table B6: Geochemical Processes (after Perkins, et al, 1995)

Mass-transfer processes	Rate-controlling processes	Rate-modifying factors
DISSOLUTION/ PRECIPITATION By: acid-base reactions Hydrolysis Redox reactions Co-precipitation Gas release/capture Wetting-drying ION EXCHANGE/ SORPTION RADIOACTIVE DECAY	DIFFUSION Macroscopic Microscopic Atomic-scale NUCLEATION SURFACE REACTION ADSORPTION/ DESORPTION	CATALYSIS Bacterial Galvanic Abiotic TEMPERATURE PRESSURE SURFACE

The precise aspects that need to be considered will vary from site to site and will depend on the questions that need to be answered. As discussed in subsequent sections, different geochemical models have different capabilities in considering the above parameters. Knowledge of the important parameters governing water quality, coupled with a knowledge of the limitations and capabilities of the different models will enable the skilled practitioner to choose the right model for each application.

B3.2.2 Data requirements for geochemical modelling

An additional important requirement for quantitative geochemical modelling is accurate and complete sets of data. Input parameters include water composition, mineralogy, bacterial activity, reactive surface area, temperature, oxygen availability, water availability, waste rock dump structure and composition, humidity and column tests, and thermodynamic and kinetic data (Morin et al., 1991).

Water chemical analyses should include acidity or alkalinity, pH, metals released from sulphide minerals (e.g. Fe, Cu, Zn, Pb, Mn, Cd), other important cations (K, Na, Ca, Mg), sulphate, other anions (Cl, S), Eh and a redox pair (Fe(II)/Fe(III)). If there is internal redox equilibrium in the aqueous phase, then measurement of either Eh or a single redox aqueous ion pair is sufficient to affect the redox state of the aqueous phase and to allow simulation of redox in the heterogeneous system. If there is redox disequilibrium in the aqueous phase, then as many redox pairs as possible should be measured. Commonly missing from acid mine water analyses are total inorganic carbon (TIC) and dissolved silica (SiO₂). These components should be determined because they are important to defining the rates of carbonate and aluminium silicate mineral dissolution. It is these

dissolution reactions that neutralize the acids formed by oxidation of sulphide minerals.

The primary acid-producing and acid-consuming minerals must be clearly and quantitatively identified in the acid rock waste dumps. Determination of secondary mineral phases is desirable but not always feasible in waste rock, typically due to the occurrence of these minerals as thin films on primary phases and/or due to their fine grain size or amorphous character. Gangue minerals may also affect the water composition. The reactive surface area of all these minerals is an important input parameter, but is one of the most difficult parameters to estimate; one usually relies on BET or grain volume measurements. Better characterization of mineral surfaces is needed as noted by Pratt et al. (1994) and Anbeek (1993).

The structure and compositional heterogeneity of the waste rock dump have major impact on the fluid flow paths and determine the location of the acid generation sites within the dump (Ritchie, 1994; Gelinis et al., 1991). Both water and oxygen must be transported into the dump to generate the acids and mobilize the metals. The supply of oxygen to the interior of a dump is mostly through gaseous diffusion and advection of air whereas the water originates as rainwater entering from the top of the dump and groundwater entering the basal portions of the dump. Both unsaturated and saturated zones are therefore observed. In addition to two-phase flow of air and water, the waste rock dumps form a complex permeability network because of the large range in grain size and heterogeneity of the dump. Heat is generated by exothermic oxidation reactions within these dumps due primarily to bacterial activity. Consequently temperature and bacterial distribution in the dump can be quite complex, but need to be documented.

Thermodynamic data are needed for secondary mineral phases. Activity coefficient models (i.e. Pitzer versus ion association) are inaccurate for concentrated solutions of aqueous iron. Fundamental kinetic data are sparse. Both surface reaction control, diffusion control, inhibitive and catalytic effects (especially the effect of bacteria) should be measured and included in the rate equations. Until this is done, reliance must be placed on field monitoring data as well as on humidity cell and column tests to calibrate empirical equations that describe metal release.

In spite of these uncertainties and incomplete or missing thermodynamic data, it is still important to collect and interpret data for use in acid rock drainage studies. This information should include field and column data, water analyses and mineralogical determinations, as outlined above. The exact ions to be determined in the water will depend on the mineralogy of the associated rock. Good data sets will always stand the test of time, and they can be used repeatedly as standards for testing geochemical models as they evolve.

B3.2.3 Uncertainties in geochemical modelling

Key uncertainties that need to be considered when undertaking geochemical modelling are:

- Mine site mineralogy (sulphide in fines), presence and nature of neutralising minerals (carbonates and aluminosilicates) and presence and nature of secondary minerals (stored acidity and sorption).
- Proportion of easily leachable (pre-oxidised) metals and radionuclides
- Water flows and quality (infiltration, pore water, seepage)
- Changes in particle size distribution over time
- Porosity and moisture content
- Oxygen transport (convection and diffusion)
- Kinetics of sulphide oxidation

It is recommended to perform deterministic sensitivity analyses to examine the influence of the various input parameters and/or assumptions. It is also proposed that probabilistic modelling be undertaken in most cases (particularly where there is significant uncertainty) in order to define the boundaries of uncertainty.

B3.2.4 Overview of different types of geochemical models

Geochemical computer modelling codes can be divided into eight classes:

- (1) Geochemical Database Generators,
- (2) Geochemical Mass Balance Models,
- (3) Geochemical Phase Diagram Generators,
- (4) Geochemical Aqueous Equilibrium Models,
- (5) Geochemical Mass Transfer Models,
- (6) Geochemical Mass Transfer-Flow Models,
- (7) Engineering Models And
- (8) Empirical Models.

The names of representative codes from each of the classes are tabulated below although individual codes are not discussed due to space considerations. All of the codes are PC-based and are described in more detail and rated individually in the MEND report by Perkins et al. (1995). The following section considers the differences between the classes and their uses in forecasting acid rock drainage.

B3.2.4.1 Class 1: Geochemical Database Generators

Most geochemical modelling programs use thermodynamic data in one form or another. Literature data are rarely in the appropriate format, and need to be manipulated or extrapolated for different conditions (e.g. temperature and pressure). The geochemical database generation models are designed to carry out these manipulations and extrapolations for the standard state thermodynamic properties of gasses, liquids and solids; they are often incorporated into other classes of thermodynamic models. A weakness is that minerals critical to the evaluation of acid rock drainage may be missing from these database generators.

SUPCRT92 (U. California at Berkeley)	GEOTAB (U. British Columbia)
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B3.2.4.2 Class 2: Geochemical Mass Balance Models

Mass balance programs, using mainly compositional constraints to calculate a solution, are among the few geochemical programs that do not depend primarily on thermodynamic data. Mass balance programs generally solve for the chemical reactions that occur between fluids and minerals based on compositional

constraints (i.e. bulk composition, mineral compositions and fluid compositions) and on hydrologic information at the initial and final points of the system. Based upon personal insight, the user selects a set of phases that may have participated in the reaction and the program then calculates the amount of each that may have been involved. This process has been termed "inverse" modelling (Parkhurst and Plummer, 1993) because these models require data input from both the beginning and the end of the reaction path or process in order to evaluate reactions occurring between these two points. In contrast, "forward" modelling (represented by model classes 5 and 6) uses only an initial compositional condition to predict the chemical composition of water and rock as a function of time. Mass balance models have little predictive capability and are mainly used to constrain the type of reactions occurring in a particular study.

BALANCE (USGS)	NETPATH (USGS)
PHREEQC (USGS)	

B3.2.4.3 Class 3: Geochemical Phase Diagram Generators

These programs calculate and display intensive-variable diagrams. Intensive-variable diagrams plot the stability fields of minerals, usually as some combination of the gaseous species (as fugacity or concentration), the aqueous species (as activity or concentration), the temperature, the pressure and/or the Eh. In the case of log activity diagrams, water analyses can be directly plotted on these figures, displaying trends in the composition of the drainage fluid and its relation to the mineral controls in the waste rock over time.

PTX and PTA (U. British Columbia)	ACT2 and TACT (U. Illinois)
BOUNDS (Pennsylvania State U.)	PERPLEX (ETH)
THERMOCALC (Powell/Holland)	

B3.2.4.4 Class 4: Geochemical Aqueous Equilibrium Models

These programs (also referred to as equilibrium models) calculate the equilibria among the aqueous species of natural solutions. The input parameters (water compositions, temperature and pressure) are used to estimate the concentrations and activities of most of the important aqueous species in a specific water, and

to calculate the saturation indices for various minerals, thus allowing the user to evaluate dissolution and precipitation of minerals. Most of these programs permit various corrections to be made for poor or incomplete water analyses, modifying the water chemistry as a result of a user-defined reaction path, or allowing constraints such as gas partial pressure or the presence of a solid phase to control the water chemistry. Their strength is based upon rigorous treatment of the aqueous phase and the flexibility to modify the fluid composition based upon a user-defined process. Aqueous equilibrium modelling can be considered a mature technology. Any weaknesses are inherited from the uncertainties in the thermodynamic and other input data used in the equilibrium model calculations.

This type of geochemical model is the easiest to use and is therefore also most often misused by applying it in inappropriate situations. Key features of the various equilibrium geochemical models that are available are:

- These are static models of an aqueous solution.
- The models can consider and use water quality analyses, temperature and pressure.
- The model solves the equilibrium distribution of mass among various solid or dissolved species and complexes.
- Results reported as saturation indexes (SI) for various minerals.
- The models can be used to predict aqueous species and maximum metal concentrations.
- The models are flexible and are widely used.

Key limitations of equilibrium models are:

- They only consider thermodynamic controls on chemical reactions and are incapable of accurately considering reactions governed by reaction kinetics such as sulphide oxidation and dissolution of aluminosilicates.
- They cannot consider physical issues such as flow rates (water balances) or particle size effects.
- They cannot produce time-dependant data and therefore cannot produce time-concentration plots or load predictions and cannot be used to evaluate the effects of management options.

EQ3 (Lawrence Livermore Nat. Lab.)	MULTEQ (EPRI)
PHREEQE (USGS)	PHQPTZ (USGS)

WATEQ4F (USGS)	ESP (OLI Systems)
MINTEQA2(EPA)	ECHEM (EPA)
SOLVEQ (U. Oregon)	GEOCHEM (U. California at Davis)
SOLMINEQ.88 (USGS - ARC)	PATH (ARC)
MINEQL (Environmental Research Software)	SOILCHEM (U. California at Berkeley)

B3.2.4.5 Class 5: Geochemical Mass Transfer Models

Mass transfer models (also called reaction path models) are dynamic models that calculate the result of reactions between a fluid and a rock in a closed system. The input parameters to these programs are the initial fluid composition and a suite of reactant minerals with their associated masses and surface areas. As the "initial" or "reactant" solids in the rock dissolve, the water composition changes. Secondary (i.e. product) solids may precipitate from, or redissolve into the evolved solution. The amount of product solids changes as a function of time until the solid assemblage is in equilibrium with the aqueous phase. The entire process is monitored as a function of time based on rate laws governing the dissolution and precipitation of the solids. When comparing the results of these programs with field observations, changes in field water chemistry over time are required.

The most serious failing of these models is the difficulty in solving ill-conditioned matrices, which occur when small changes in the reactants result in large changes to the solution composition. Numerically following a reaction path over an extremely large change in the case of a small amount of a component approximates a discontinuity. In acid rock drainage, the oxygen "cliff" (the change from oxidizing to reducing conditions) is the most common case. The activity of dissolved oxygen changes from near atmospheric conditions to values of the order to 10^5 over very small changes in the other components. Reaction step size has to be very small at this point and consequently the simulations can take a very long time to complete, or even fail under these conditions. Modelling of dilute solutions under these conditions, when no buffering phases are present offers the greatest challenge. Additionally, key limitations of these models in geochemical assessment programmes are:

- They do not truly represent field conditions in that they do not incorporate flow through systems.

- The models are complex to use.

PATHARC (ARC)	CHILLER (U. Oregon)
EQ6 (Lawrence Livermore Nat. Lab.)	REACT (U. Illinois)
STEADYQL (Furrer, Oregon State U.)	

B3.2.4.6 Class 6: Geochemical Mass Transfer - Flow Models

Mass transfer-flow models (also referred to as mass transfer-mass transport models, reaction path-flow models, reactive flow models, or reaction path-mass transport models) are essentially open-system versions of mass transfer models; that is, they are mass transfer models that have been expanded to include flow. These programs are complex and are aimed at simulating a wide variety of geochemical processes. In addition to flow, these models consider solute transport and some consider heat transport. However, the complexity of the programming has severely limited the number of "general" programs of this class, and compromises have been made. Some models focus on the geochemistry of the system, while others emphasize hydrologic aspects.

The traditional approach to solving this complex problem (generally taken by people with a background in groundwater hydrology or reservoir modelling) has been to develop or use an existing 2-D or 3-D fluid flow code and to include in it transport equations for one or more chemical components. The components are often inert or react with the solid matrix through a simple distribution coefficient. In recent years, advances in these models have been driven both by the needs of the oil industry to predict thermal-enhanced oil recovery (or similar processes) and by groups concerned with predicting the fate of subsurface contaminants. The complexity of the reaction terms and the number of components have increased in some models, while in other models the ability to deal with thermal events (temperatures up to 350°C) and flashing/condensation has been included.

An alternative approach has been taken by those with a background in geochemistry. These individuals concentrated their efforts in describing the geochemical reactions that occur between the fluid and the minerals, using only the minimum information necessary to describe the fluid flow, heat flow and solute transport regimes. The geochemical portions of these codes are essentially open-system versions of the mass transfer models described above. The methods used to couple

the flow and transport equations to the geochemical equations vary from model to model, and primarily depend on the class of problem that the author intended to address.

Data required for these programs include detailed water analyses describing the input composition as a function of time, the initial mineralogy over the area being covered, the flow rates into or out of the system, and temperature changes over time.

The latest programs in this category, e.g. HYDRO-GEOCHEM, allow for the simulation of reactive multispecies-multicomponent chemical transport through saturated-unsaturated media. HYDROGEOCHEM is not a path model, instead it is a true transport model coupled with homogeneous and heterogeneous geochemical reactions. It is designed to simulate transient and/or steady-state transport of aqueous components and transient and/or steady-state mass balance of adsorbent components and ion-exchange sites. The model is designed to.

- (1) Treat heterogeneous and anisotropic media,
- (2) Consider spatially and temporally-distributed as well as point sources/sinks,
- (3) Accept the prescribed initial conditions or obtain initial conditions by simulating the steady-state version of the system under consideration,
- (4) Deal with prescribed transient concentrations distributed over a dirichlet boundary,
- (5) Handle time-dependent fluxes over variable boundaries,
- (6) Deal with time-dependent total fluxes over cauchy boundaries,
- (7) Include the off-diagonal dispersion coefficient tensor components in the governing equation for dealing with cases when the coordinate system does not coincide with the principal directions of the dispersion coefficient tensor,
- (8) Provide two options for treating the mass matrix - consistent and lumping,
- (9) Give three options (exact relaxation, under- and over-relaxation) for estimating the non-linear matrix,
- (10) Include two options (direct solution with gaussian elimination method and successive point iterations) for solving the linearized matrix equations,

- (11) Include both quadrilateral and triangular elements to facilitate the discretization of the region,
- (12) Automatically reset time step size when boundary conditions or sources/sinks change abruptly, and
- (13) Include simultaneous chemical processes of aqueous complexation, precipitation/dissolution, adsorption, ion exchange, redox, and acid-base reactions.

Key limitations of these coupled mass transfer - flow models are:

- The complexity of the models can lead to a compromise between hydrology and geochemistry.
- The complex programming limits the general application of these models and they can only be confidently operated by geochemical specialists.

PHREEQM (USGS)	FMT (C. Novak, Sandia Nat. Lab.)
REACTRAN (C.H. Moore, Indiana)	KGEOFLOW (D. Sevougian, Battelle)
CIRFA (P. Ortoleva, Indiana U.)	IDREACT (C. Steefel, Battelle)
MPATH (P. Lichtner, Southwest Research I.)	FCT (C. Steefel, Battelle)
Unnamed (W. White, USBM)	MINTRAN (Frind, U. Waterloo)
UNSATCHEM (Suarez, USSL)	HYDROGEOCHEM (Yeh & Salvage, Penn State)
ROCKSTAR (SENES)	

B3.2.4.7 Class 7: Engineering Models

Empirical/engineering models do not perform general geochemical calculations, but instead describe a specific situation; for this reason they are here termed applied geochemical models even though the geochemistry included in them may be quite limited. The number of geochemical and physical processes is limited as much as possible without losing the capability of simulating the bulk geochemical and physical behavior of the event. This class is distinguished from the other six classes of geochemical models because the other classes are, for the most part, general in nature and may be applied to many different geochemical problems without modification in addition to acid rock drainage.

In engineering models, the geochemical processes that are included are only those which are assumed to significantly influence gross acid rock drainage changes over time. The geochemical processes are represented in the models by sophisticated kinetic rate laws for dissolution and precipitation of the important solid phases. Homogenous equilibria in the aqueous phase is assumed.

At the other extreme of the applied geochemical models lie those empirical models that treat the geochemistry and the physics in the simplest terms. An example is the model of Morin and Hutt (1994) which ignores many of the complexities of both geochemical and physical processes and uses only five mass balance and time factors to predict acidic seepage chemistry through time from waste rock dumps.

Key features of the various engineering models can be summarised as:

- They simulate bulk geochemical and physical behaviour of waste deposits.
- They can incorporate the results from kinetic laboratory studies.
- They use simplifying assumptions (e.g. uniform properties within nodes, simple geometry) and have simplistic geochemical routines, although they generally have a capability of considering both thermodynamic and kinetic reaction controls.
- They are primarily useful for examining the effects of dominant processes and to compare alternative management, rehabilitation and decommissioning options.

Key limitations of these models relate to their simplified consideration of geochemical processes.

Examples of engineering models are WATAIL, RATAP, ACIDROCK, FIDHELM, Q-ROCK, MINEWALL.

WATAIL (U. Waterloo)	RATAP (CANMET)
Q-ROCK (SRK)	ACIDROCK (SENES)
FIDHELM (ANSTO)	MINEWALL (Morin)
THE GEOCHEMIST'S WORKBENCH (Bethke, U. Illinois)	

B3.2.4.8 Class 8: Empirical Models

A separate class of geochemical models that could be identified are the empirical / statistical models and they

have a significant role to play in geochemical assessments and prediction of future pollution from mining sites. They are, however, not applicable in all cases as their prediction of future water chemistry is based on an extrapolation from current conditions. In cases where the change of key variables affecting water quality with time is easily predictable and the effects of these changes on water quality is well understood, empirical models can be used to give good estimates of future water quality.

The basic procedure that is followed with empirical modelling and prediction is:

- 1) Examine a water chemistry database (either from field or kinetic laboratory measurements) and assess the adequacy of the data and remove outliers.
- 2) Compile summary statistics for the data.
- 3) Make use of uni or bivariate assessment techniques to compile time plots and undertake correlation and regression analyses.
- 4) Make use of multivariate assessment techniques such as principal component analyses, cluster, factor or discriminant analyses.
- 5) Assess cyclical or repeating trends and undertake time series modelling.
- 6) Develop an empirical model and cross-validate it with additional data sets or simple modelling techniques to assess the confidence and reliability of the model.

Geochemical processes not explicitly incorporated within the empirical model are often evaluated through the application of equilibrium speciation models. On the other hand, empirical models can often be included within the framework of mechanistic models using the following example process:

- 1) Analyse results obtained from kinetic testwork (e.g. humidity cells, columns).
- 2) Develop an empirical expression relating constituent concentration to pore water replacement volume.
- 3) Apply this expression within a mechanistic framework (e.g. flow and mass transfer model)
- 4) Perform a geochemical assessment using the mechanistic model.

B3.2.4.9 Applicability of Different Geochemical Models

The first six model classes, described above, are also in order of increasing complexity. The model representatives from the seventh class (empirical/engineering models)

vary in complexity, from the simple empirical models to the complex engineering models. Only the last four classes of models have predictive capability and are therefore of most interest for use as an aid in acid rock drainage decision-making. Engineering models require the largest number of input parameters. However, mass transfer models, which address geochemistry in more detail, require more geochemical data.

Aqueous equilibrium models have been used to identify the soluble and mobile metal species, their maximum concentrations and their relations to the minerals in the mine waste rock, and are widely used for acid rock drainage studies (see Blowes and Jambor, 1990, and other examples in Jambor and Blowes, 1994).

Mass transfer models have seen few applications in acid rock drainage. They should be considered for use in Acid Base Accounting (ABA). Support for this lies in the observations of Kwong (1993) on weatherability of common sulphide minerals and the advocacy of R. Lawrence for the use of normative mineral analyses for understanding the complexities of ABA.

In addition to oxidation, dissolution and precipitation, mass transfer models can also simulate evaporation, which is another important process in the unsaturated zone of waste rock dumps. One combined effect of oxidation, dissolution and evaporation in acid rock drainage is to increase the concentration of those ions in the aqueous phase that can result in the precipitation of highly soluble sulphate minerals. In acid rock drainage, evaporation is normally followed by a dilution event as more water percolates through the dump, resulting in the dissolution of the sulphate minerals. This cycling of the secondary sulphate precipitates may control the TDS of some of the waters during flushing events.

Flushing is mainly a transport event and requires integrating flow into the mass transfer models; i.e. they become mass transfer-flow models. Obviously, it is difficult to decide on a unique path for these events unless one knows the rate of evaporation and the subsequent rate of flushing of the water upon addition of the next increment of water. Perkins et al. (1995) demonstrated that mass transfer modelling of a column experiment correctly simulated the formation of acid saline water from neutral fresh water, although the absolute concentrations of the ions in the saline water were not matched in detail. This provides hope that more complex calculations could lead to a relatively accurate simulation, provided that the physics of the flow is known exactly and is incorporated

into the mass transfer-flow models. Mass transfer-flow models have the best potential for predictive capability in the long term. Unfortunately, both these classes (mass transfer and mass transfer-flow) have rarely been used in acid rock drainage research. The use of mass transfer-flow models for such applications is not recommended because none of them currently addresses both the physical and geochemical aspects of acid rock drainage well enough. As a first step mass transfer models should undergo more rigorous testing in acid rock drainage applications.

Empirical/engineering models aim at the prediction of acid generation and metal loadings under different containment conditions and physical structures. They can be useful for comparing various decommissioning options. However, these models are best at history matching. They do not include detailed geochemistry and thus have limited predictive capacities.

A summary of the differences in data requirements between the different geochemical models are shown in Table B7.

Table B7: Data Requirements of Different Geochemical Models

Input	Parameters	MODEL CLASS			
		EQUIL.	M.T.	M.T./FLOW	EMP/ENG
Field Data	Water Chem.	+++	++	++	+
	Mineralogy	+	+++	++	+
	Surface Area	0	+++	+++	+
	Temperature	+	+	+	+
	Oxygen	+	++	++	++
	Water Balance	0	+	++	++
	Dump Structure	0	0	0	++
Lab Data	Column Test	0	0	0	+
	Humidity Cell	0	0	0	+
Database	Thermodynamic	+++	+++	+++	++
	Kinetic	0	+++	+++	+

"Equil." = aqueous equilibrium models,

"M.T." = mass transfer models,

"M.T.-Flow" = mass transfer-flow,

"Emp./Eng." = empirical/engineering models.

Symbols expressing relative importance of input parameters are: "0" = none or not required, "+" = required the least, "++" = intermediate requirement, "+++ required with most accuracy.

Geochemical models have been grouped into eight classes based on the nature of their application to acid rock drainage studies: Database generators create the necessary thermodynamic data input for classes three to six. Phase diagram generators are useful in displaying trends in the composition of the drainage fluid and its relation to the mineral controls over time. Mass balance models compare aqueous influx and outflux concentrations and mineralogy to solve for a bulk chemical reaction between the fluid and minerals. These models have little or no predictive capability.

Aqueous equilibrium models are static models used to identify the soluble and mobile metal species, their maximum metal concentrations, and their relation to the minerals in the mine waste rock. Mass transfer models are dynamic reaction path models for closed systems that address maximum metal concentrations and their evolution with time. Mass transfer-flow models are dynamic reaction path models in an open system that addresses the prediction of concentration, load and distance travelled over time. Empirical/engineering models are best at history matching and have limited predictive capability. They are more appropriate for examining decommissioning options in acid drainage studies. More developments are required in each model category to achieve each prediction objective in a

satisfactory manner.

Existing geochemical models should not be used with the objective of predicting exact water chemistry from mining features. Rather, they should be used to improve the understanding of the interactions between geochemical processes and to perform comparisons between decommissioning scenarios. Better input data for these models and better linking between geochemical and physical processes in the models are needed. Empirical/engineering models based on laboratory and field tests must be used and further developed, in the short term, to compensate for the limitations of presently available geochemical models.

Based on the capabilities and limitations of the different types of geochemical models, Perkins et al (1995) have recommended the applicability of the different models based on the objectives of the prediction as summarised in Table B8.

Table B8: Model Applicability Versus Prediction Objectives

PREDICTION OBJECTIVE	MODEL CLASS			
	EQUIL.	M.T.	MT/FLOW	EMP/ENG
ID soluble & mobile species	+++	++	+	0
Predict max. concentrations	+	++	+	0
Predict max. loads	+	++	++	+
Predict duration	0	++	+++	++
Predict concentration vs time	0	+	++	+
Evaluate/compare decommissioning options	0	0	++	+++

"Equil." = aqueous equilibrium models,

"M.T." = mass transfer models,

"M.T.-Flow" = mass transfer-flow,

"Emp./Eng." = empirical/engineering models.

Symbols expressing relative importance of input parameters are: "0" = none or not required, "+" = required the least, "++" = intermediate requirement, "+++" required with most accuracy.

**APPENDIX C
CONSIDERATIONS
FOR INCORPORATION
INTO CONCEPTUAL
MODELS**

Broadly speaking, mining features that are typically associated with impacts on the water resource can be divided into the following four categories:

- i) Coarse mine residue deposit (coarse discard dump, waste rock dump, spoils heap)
- ii) Fine mine residue deposit (tailings disposal facility, coal slurry dam)
- iii) Opencast pit (need to distinguish between the following stages of a pit: open or backfilled pit; unflooded; partially flooded; flooded; operational; defunct)
- iv) Underground mine (need to distinguish between the following stages of an underground mine: unflooded; partially flooded; flooded; operational; defunct)

Each of these mining features are discussed in more detail in order to understand how they function in terms of the physical, chemical and biological driving forces that affect mine water quality.

C1 COARSE MINE RESIDUE DEPOSITS

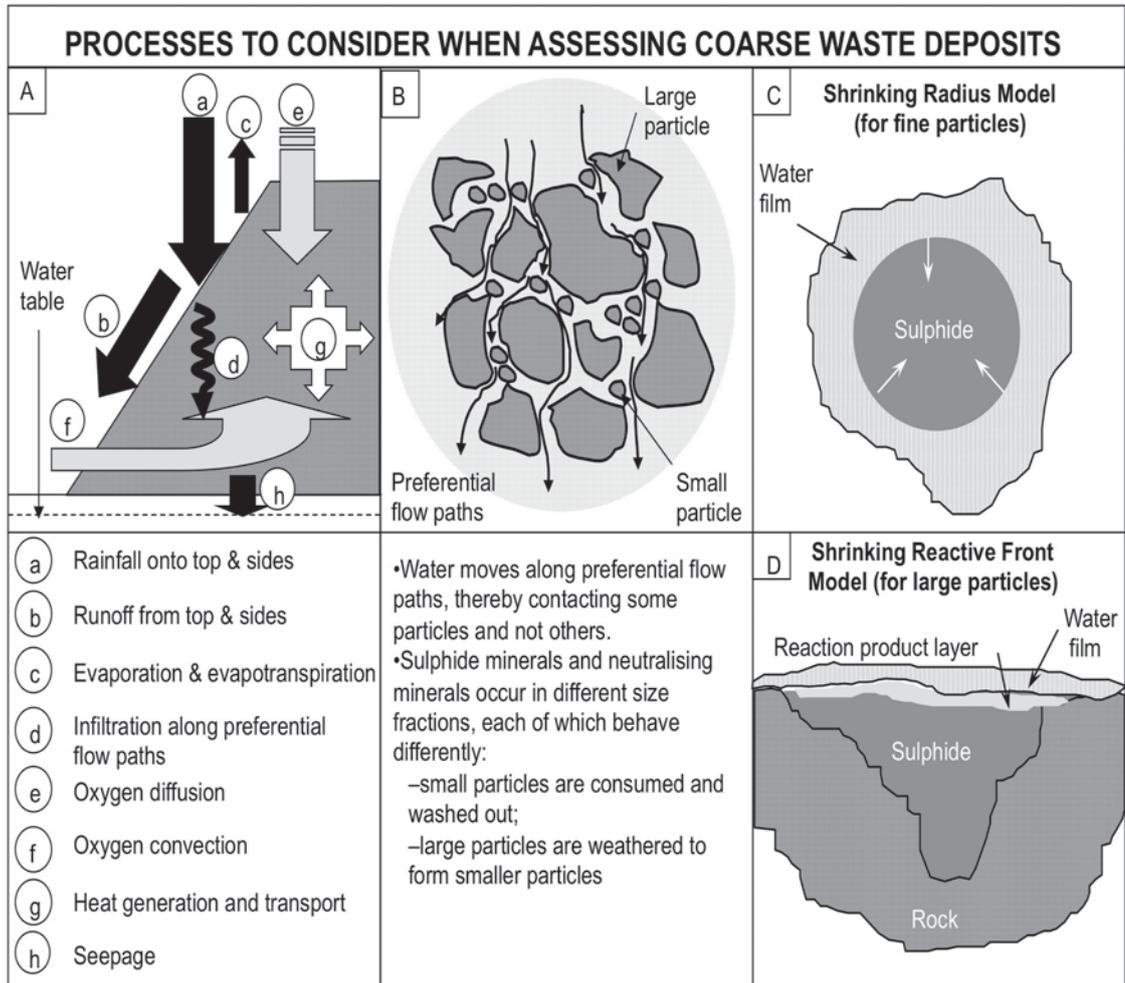
Coarse waste residue deposits typically provide an environment that is very conducive to the production of acid rock drainage. This is primarily due to the fact that coarse waste deposits are very permeable and allow the ingress of oxygen to considerable depths. However, this is counteracted by the fact that, per definition, coarse waste has a large particle size, thereby ensuring that a large amount of the minerals that contribute to the change in water quality, are locked up within the particles.

The driving forces and processes that characterise the geochemical behaviour of a coarse waste residue deposit and that need to be explicitly considered when assessing the quality of water draining from such a waste deposit are shown schematically in Figure C1 below.

Figure C1-A shows that the following macro-scale driving forces need to be considered in a coarse waste residue deposit:

- Rainfall onto the top and sides of the dump
- Runoff from the top and sides of the dump
- Evaporation and evapotranspiration from the top and sides of the dump
- Infiltration into the dump along preferential flow paths
- Diffusion of oxygen into the dump
- Convection of oxygen into the dump
- The generation of heat within the dump (from sulphide mineral oxidation and/or spontaneous combustion [coal discard]) and the transport of this heat into and out of the dump
- Seepage of contaminated water from the dump (either at the toe or through the base)

Figure C1: Processes to Consider when Assessing Coarse Mine Deposits



In Figure C1-B, a close-up view of the inside of the coarse waste residue dump is shown. An important feature that needs to be considered is that the dump will typically contain particles over a fairly wide size range. The small particles are more rapidly consumed, either through sulphide oxidation (chemical and/or biological), neutralisation (chemical) or washout through the dump (physical). The larger particles, in turn, are weathered to form and replenish the inventory of smaller particles.

The phreatic surface within the coarse waste residue deposit is dependant on the volume of water ingress on the dump surface and the permeability of the underlying weathered material/aquifer relative to that of the waste material. It is anticipated that the underlying lithologies generally have a lower permeability than that of coarse waste residue deposits.

Since the permeability of the coarse waste material is relatively high, one would expect the phreatic surface within the dump to be relatively flat and located within the lower portion of the dump. This depressed water table has two very important effects that need to be considered:

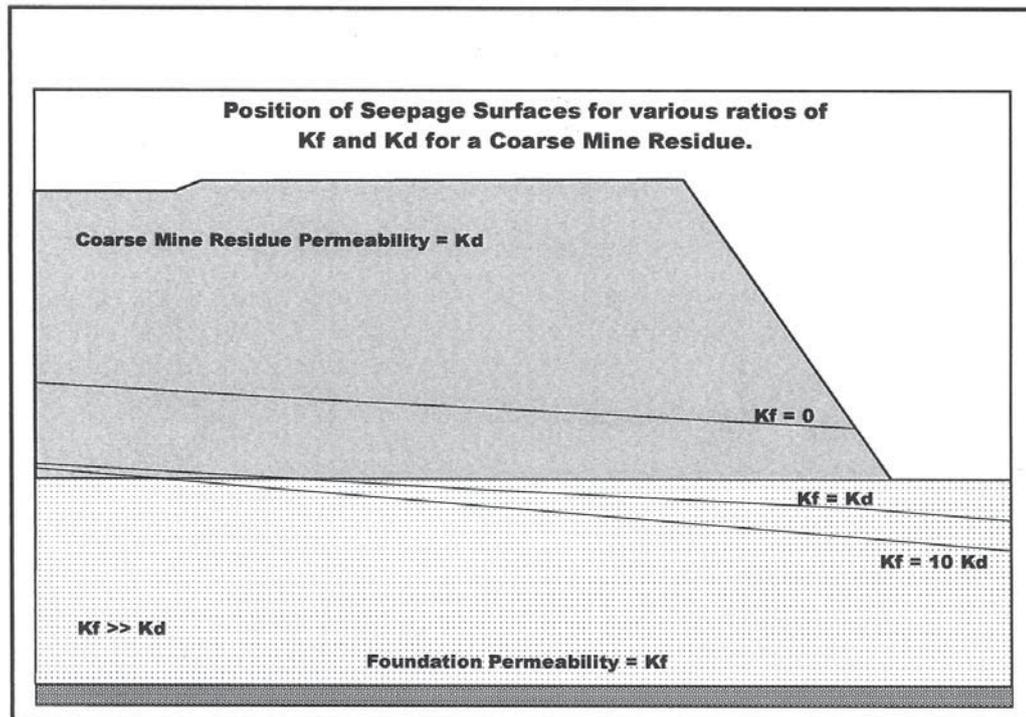
- Water flows vertically down through the dump along preferential flow paths - this means that some particles are fairly isolated from the water flow path.
- Oxygen penetrates deep into the dump and the spaces between particles are typically filled with air (and oxygen), while particles are surrounded by a thin layer of water.

Leachate from a coarse waste residue deposit is most likely to seep from the face of the dump in close proximity

to the toe rather than percolate into the underlying aquifer. This is attributed to the relatively low permeability of the underlying material which cannot accommodate the seepage volume from the coarse waste residue for any given rainfall event. This is counteracted by the fact that the mass of the dump will typically create a depression of the soil under the dump, allowing seepage to accumulate

within the depression. The potential groundwater gradients associated with coarse mine residues are shown in Figure C2.

Figure C2: Potential Groundwater Gradients Associated with Coarse Mine Residues



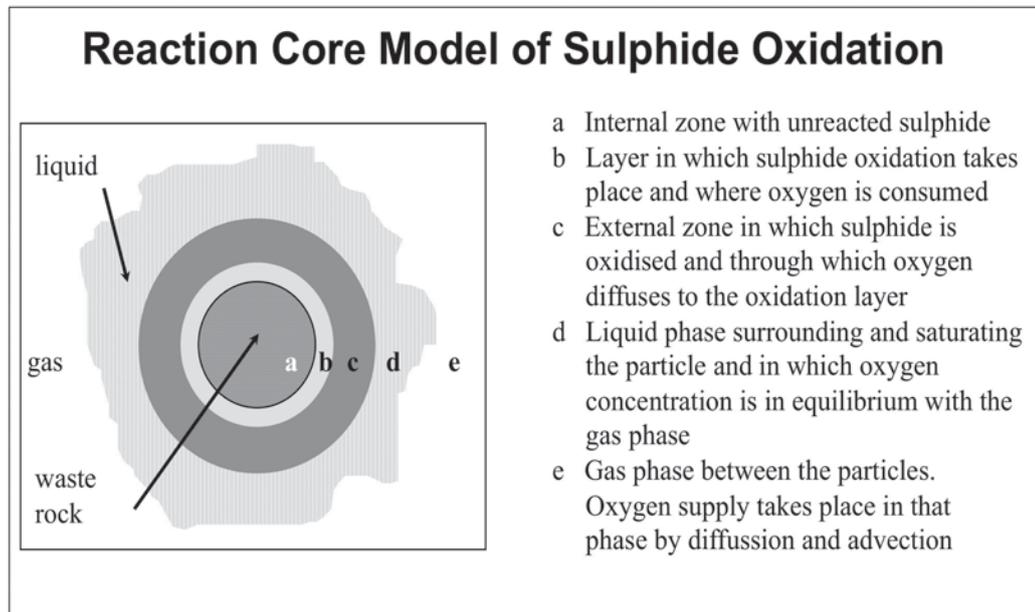
Uncontrolled leachate that emanates from the coarse waste residue as surface runoff will flow downgradient until it percolates into the weathered soil profile. The distance that such a surface runoff covers before seeping into the underlying aquifers is dependant on the seepage volume, the permeability of the underlying material and the topographic slope in the immediate vicinity of the coarse waste residue. The vertical percolation of the leachate into the underlying aquifers may be hindered by the upward convergence of groundwater in the immediate vicinity of water courses. However, relatively clean groundwater will come into contact with mine waste residue as a result of the convergence process. This contaminated groundwater will usually enter the water courses as base flow.

The fine and coarse particles can be considered to behave differently, as shown in Figures C1-C and C1-D.

Whereas a very fine particle could be modelled as only containing reactive material (sulphide or basic mineral) exposed on all surfaces, the coarse particle will contain reactive material embedded within the host rock and only a portion of the particle will be reactive.

As shown in Figure C1-C, the fine particle could be modelled with the shrinking radius model where it is consumed by the reaction, thereby growing smaller over time until it is depleted. Alternatively, the particle could be modelled with the reaction core model shown in Figure C3, where various micro-scale physical processes are considered.

Figure C3: Reaction Core Model of Sulphide Oxidation



For the larger particle, as shown in Figure C1-D, only a portion of the reactive mineral surface is exposed. This surface is coated by a layer of reaction products, followed by a thin film of water.

At a microscopic level, it is found that different sulphide minerals and even different forms of the same mineral, have different structures that affect their reactivity. For example, certain forms of pyrite have a porous structure that allow oxidation to occur within the particle, while other forms of pyrite are "glass-like" with a highly impermeable structure and oxidation will only occur on the outside surface.

Finally it has been established, by many researchers, that the process of sulphide oxidation within an oxygen-rich environment, such as occurs within a coarse waste residue deposit, is both a chemical and a microbiological one. Although there are different theories as to the precise role of the microbes, it is well established that these microbes can increase the rate of the sulphide oxidation reaction by a factor of between 10 and 1000 times.

C2 FINE WASTE RESIDUE DEPOSITS

Fine waste residue deposits exhibit a number of very important differences when compared to coarse waste residue deposits:

- Particles are relatively uniform in size and, at the very least, exhibit a much smaller size distribution.
- The water table is typically elevated with a large portion of the dump being in a saturated state (in operational facilities, the water table may be at the surface of the dump).
- The relative proportion of seepage emanating from the toe of the dump relative to that seeping into the underlying aquifer is dependant on the permeability contrast between the waste material and the surrounding lithologies.

As a result of these differences, the environment within the dump is less conducive to the production of acid rock drainage. This is primarily due to the fact that the penetration of oxygen is limited (mainly true for operating facilities receiving large regular inputs of water). However, this is counteracted by the fact that fine waste has a small particle size and, consequently, a large surface area of reactive minerals. Additionally, fine wastes are typically hydraulically placed and the water medium itself may be a major source of contaminants.

The driving forces and processes that characterise the geochemical behaviour of a fine waste residue deposit and that need to be explicitly considered when assessing the quality of water draining from such a waste deposit are shown schematically in Figure C4 below.

Figure C4: Processes to Consider when Assessing Fine Mine Deposits

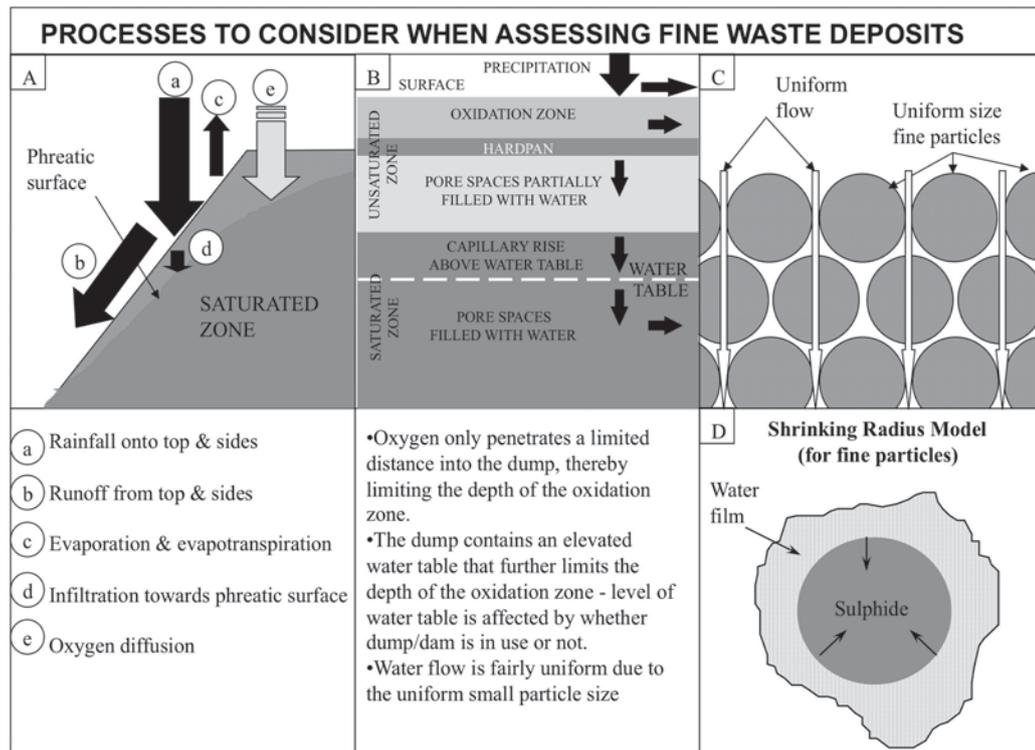


Figure C4-A shows that the following macro-scale driving forces need to be considered in a fine waste residue deposit:

- Rainfall onto the top and sides of the dump
- Runoff from the top and sides of the dump
- Evaporation and evapotranspiration from the top and sides of the dump
- Infiltration into the dump along preferential flow paths towards the phreatic surface (water table)
- Diffusion of oxygen into the dump
- Seepage of contaminated water from the dump (either at the toe or through the base)

In Figure C4-B, a close-up view of the inside of a fine waste residue deposit is shown. There are a number of important features shown here that need to be considered when evaluating the driving forces that occur:

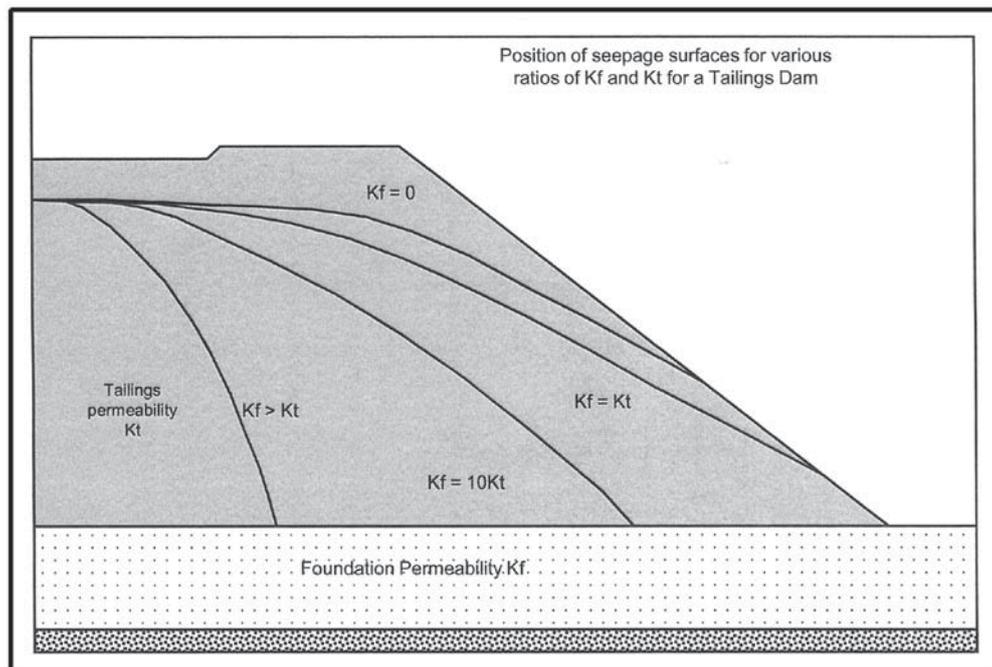
- The dump consists of an unsaturated upper zone and a saturated lower zone.
- The unsaturated zone can typically be subdivided into three separate zones:
 - an uppermost oxidation zone where the bulk of the sulphide oxidation will occur;
 - a hardpan that effectively reduces the amount of vertical oxygen and water penetration and results in the lateral movement of infiltration water; and
 - a lower zone where the pore spaces are partially filled with water and where sulphide oxidation occurs at a reduced rate.
- The saturated zone typically extends somewhat above the true water table due to the capillary rise of water into the pore spaces. This saturated zone does not allow for any significant sulphide oxidation

although effective neutralisation by base minerals may still occur here.

Figure C4-C shows that, compared to a coarse waste residue deposit (Figure C1-B), a much tighter packing density will occur due to the more uniform particle size resulting in more uniform flow, particularly in the saturated zone. The sulphide oxidation process could be modelled either by the shrinking radius model (Figure C4-D), or the reactive core model (Figure C3).

Due to the above features and the more uniform processes that occur, fine waste residue deposits are generally easier to model reliably than coarse waste residue deposits. The potential variation in groundwater gradients associated with fine mine residues are shown in Figure C5.

Figure C5: Potential Groundwater Gradients Associated with Fine Mine Residues



C3 OPEN CAST PITS

The processes that control the water quality in an open cast pit are very dependent on the nature of the pit, with the following parameters being particularly important:

A Pit Backfill Status

- A.1 Open pit - no backfill of material
- A.2 Open pit - partially backfilled with open sections or ramps
- A.3 Backfilled pit

B Nature of Backfill Material

- B.1 Backfilled with material that does not contain any sulphide or reactive material

- B.2 Backfilled with coarse material that contains potentially reactive material

- B.3 Backfilled with fine material that contains potentially reactive material

C Position of Backfill Material

- C.1 Potentially reactive material above the water level
- C.2 Potentially reactive material below the water level
- C.3 Potentially reactive material protected by an oxygen or infiltration barrier above, below or around the material

D Hydrological Status of Pit

- D.1 Pit completely unflooded
- D.2 Pit partially flooded

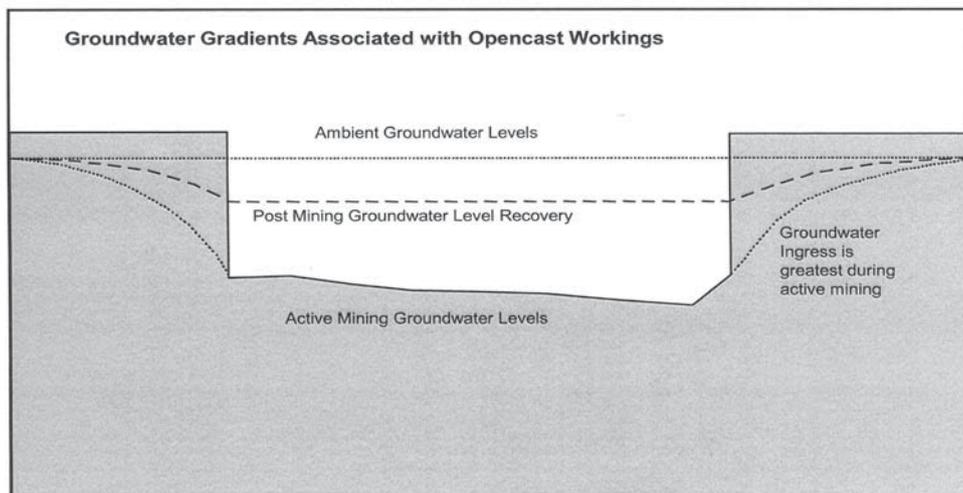
- D.3 Pit completely flooded
- D.4 High ground water or surface water flow through the pit
- D.5 Low ground water or surface water flow through the pit
- D.6. Base of the Pit above the regional groundwater elevation

All open cast pits will progress through different phases during their life and it is important that this aspect be considered whenever assessing the geochemical status of a pit. The previous history of the pit plays an important role in defining the future behaviour of that pit. For example, a pit that is backfilled, completely flooded and currently decanting, will possibly be re-dissolving secondary minerals that were formed and stored in the pit during the time that it was only partially flooded and active sulphide oxidation was occurring. Therefore, while sulphide oxidation may currently not be occurring, the fact that it did occur previously may affect future water quality.

The features within an open cast pit that are potential sites for geochemical reactions are the pit walls and floor and the backfill material. The walls and floor are normally less important due to their relatively small surface area compared to the backfill material. The different components of an open cast pit can generally be likened to the features of the coarse or fine waste residue deposits, with the exception that in a pit, the coarse material could also be located either above or below the water table.

The additional critical feature to consider in a pit is the influence of ground water movement into and through the pit. In the initial stages following the cessation of mining, the groundwater gradients will be at their steepest in the direction of the pit unless there is some interconnection to other workings - see Figure C6.

Figure C6: Potential Groundwater Gradients Associated with Opencast Workings

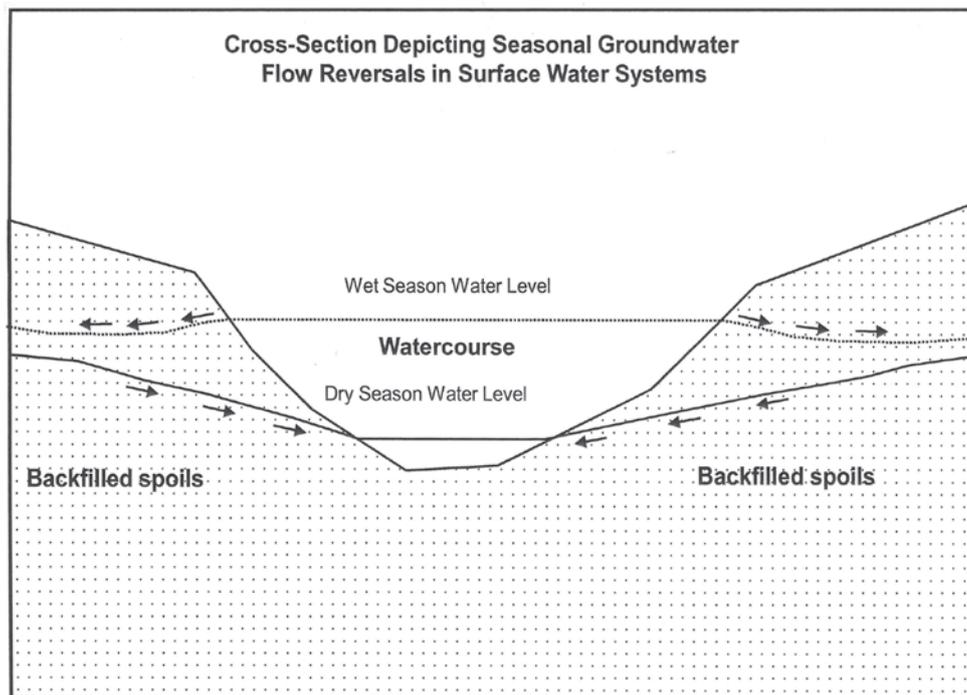


Groundwater ingress volumes will be greatest during this mining phase and immediately thereafter. This volume of water will reduce over time as the groundwater gradients become less pronounced in response to the groundwater level recovery. The groundwater through flow will be at a minimum once the surface decant is reached or the ambient groundwater gradients are re-established. A subsurface seepage from the opencast pit may occur where the depth of weathering is greater prior to the surface decant.

Surface water systems such as rivers may also have a profound effect on the water quality within opencast workings especially where the water course has been undermined and there is a direct interaction between the mine workings and the water course. A flushing phenomenon is commonly observed under these circumstances where relatively clean water seeps from the water course into the opencast workings during the wetter summer months - see Figure C7. The groundwater flow is reversed during the drier winter months as the

water level within the water course drops relative to the water level within the spoil releasing a salt load to the respective catchments. This scenario promotes ongoing geochemical reactions within the spoil since it introduces the oxygen and water required for the formation of acid rock drainage.

Figure C7: Cross-Section Depicting Seasonal Groundwater Flow Reversals in Surface Water Systems



C4 UNDERGROUND MINES

C4.1 Geochemical Considerations

From a geochemical perspective, the underground mines can be divided into the following different mining features:

- A Competent mining panels
- B Collapsed mining panels

Competent mining panels are defined as those that have experienced minimal collapse or closure and where significant mine voids still exist. As with the collapsed mining panels, the primary features of competent mining panels relate to the nature and abundance of reactive materials and the hydrological regime.

The competent mining panels are generally characterised as having smaller amounts of reactive material that can

affect water quality. There are competent mining voids, however, that have not been cleaned properly and that still contain significant amounts of reactive material, primarily of a relatively small particle size. Such "dirty" areas may be similar to or worse than the collapsed mining panels in terms of the available reactive material.

Collapsed mining panels generally have the effect of placing a large amount of material within the water flow path. In areas where the ore body has been totally removed (gold mines and high extraction coal mines), the nature of the material in the roof (hanging wall) will affect the quality of the water flowing through the panel. Where only partial ore body removal has occurred (e.g. split and quartered bord & pillar coal mines) the material in the flow path will be a combination of ore body and roof material.

The reactive material that could be found in the mining panels includes the following:

- sulphide minerals
- neutralising minerals
- secondary minerals (precipitated due to neutralising reactions on ARD)
- chemical sludges deposited underground
- fine material (coal slurry or gold tailings) backfilled into underground voids

In terms of hydrological regime, the following distinctions can be made for both competent and collapsed mining panels:

- flooded and stagnant
- flooded but with active flow paths through the panel
- moist with flowing water through the panel
- completely dry

A mining panel that is flooded and stagnant (isolated from the primary flow paths) is generally incapable of making a significant contribution to the overall water quality within the mine and can often be ignored, except for its effect on flooding and decant lag time.

A mining panel that is flooded but that has active flow paths through the panel could contribute significantly to overall mine water quality. Although the generation of contaminants will generally be insignificant due to the lack of oxygen (in some cases the flow path could include highly oxygenated water in which case sulphide mineral oxidation may well be significant), the processes of dissolution of neutralising minerals and chemical sludges as well as dissolution or precipitation of secondary minerals may continue at significant rates.

A mining panel that is moist (humid) and contains free flowing water will often be the major source of contamination as the conditions are good for sulphide mineral oxidation. In situations where the depth of the free flowing water varies with time (e.g. due to response to rainfall and recharge), portions of the reactive minerals will go through repeated wetting and drying cycles. This action will also tend to promote the oxidation of sulphide minerals. In moist humid conditions, a continuous trickle of highly contaminated seepage from the moist material into the flowing water may be the dominant source of contaminants.

A completely dry panel will, similarly to a flooded stagnant panel, have little or no impact on overall mine water quality. Although such panels may have an abundance of

oxygen, ARD cannot be generated without the presence of water.

An assessment of an underground mine will typically result in the development of a conceptual model that has a combination of collapsed and competent mining panels with different hydrological regimes. Generally speaking, an adit mine into a hillside will exhibit the most complex combination of mining panels while shallow underground mines in topographically flat areas will exhibit the least complex combination.

C4.2 Geohydrological Considerations

During underground mining operations, water ingress into the workings is usually controlled by means of pumping or by gravity drainage along adits where mining has taken place into hillsides. The quantity of water ingress is dependant on the origin of the water and the recharge sources in the immediate vicinity and the permeability of the aquifers adjacent to the mine workings.

Once the mine workings are defunct, the rate of groundwater recovery is dependant on the origin of the water influx into the workings and the storage capacity of the remaining mine voids. Whether the mine water will ultimately decant is dependant on these two factors. Groundwater contamination migration away from the underground workings is only likely to occur once the groundwater gradients have re-established themselves or a surface decant point has been reached. Predicting groundwater recovery is therefore an important component in the forecasting of groundwater contaminant migration from underground workings.

C4.3 Definition of Underground Mining Scenarios

Underground mines may be subdivided into five categories namely:

- Underground workings accessed by means of adits
- Deep Witwatersrand type gold mining operations
- Deep Bord and Pillar Coal Mining
- Shallow Bord and Pillar Coal Mining
- High extraction Coal Mining

The geochemical and groundwater regime around these mines and the resultant conceptual models will be discussed in more detail below.

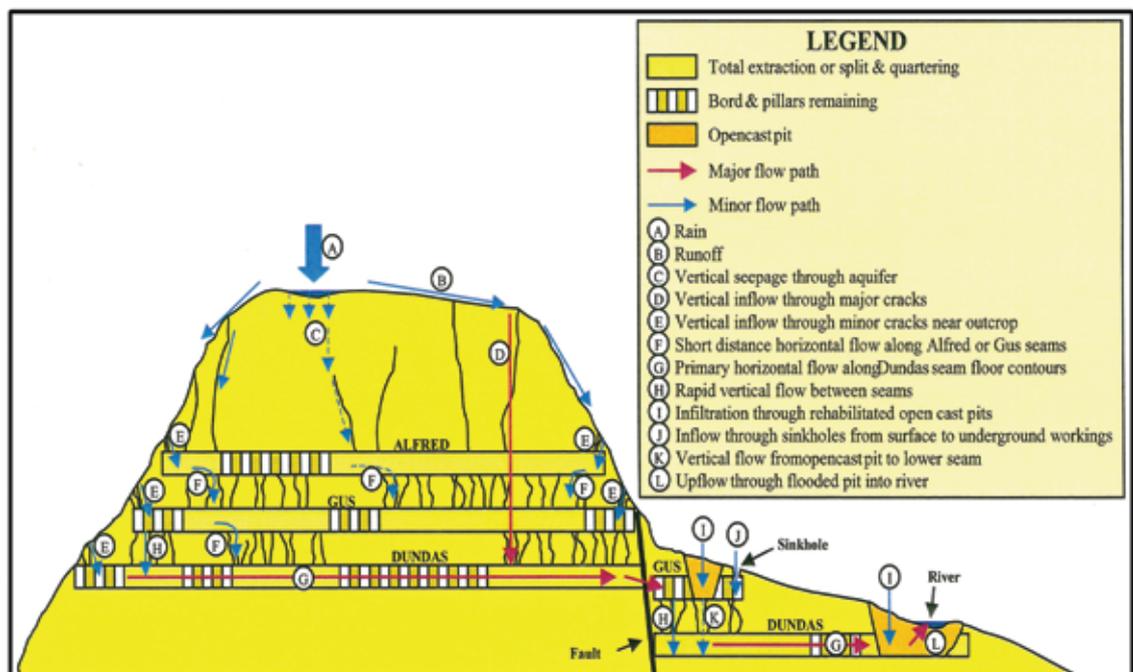
Underground workings accessed by means of adits

Adits are horizontal tunnels that are used to access ore deposits in hilly regions. This method is more cost-effective than shaft sinking since the development often occurs within the ore thereby recouping the capital cost of the initial workings. Another “advantage” of adits is that they are often sloped towards the edge of the mountain so that any excess water may freely drain from the active

working face reducing the need for additional pumping. Examples of such mines include the greenstone gold mining at Pilgrims Rest and various coal mining activities in KwaZulu Natal.

This type of mine presents a particularly complex geochemical and geohydrological problem to solve as the mine will generally exhibit active through flow and decant of water and will also contain most if not all of the possible geochemical scenarios discussed in Section C4.1 above. An example of a conceptual representation of such a mine (a coal mine in Kwazulu Natal) is shown in Figure C8 below.

Figure C8: Integrated conceptual model for an underground adit coal mine



Water ingress into these workings is the result of a number of factors namely:

- Natural percolation of water into the subsurface as groundwater recharge including ingress via geological features
- Additional recharge and surface runoff into mining induced cracks particularly along the edges of the mountains
- Recharge into subsidence areas where the workings are close to the base of the weathering

Although groundwater ingress may occur during the initial stages of mining, this source of water diminishes

over time as the overlying strata are dewatered. Water ingress into these workings is therefore dependant on rainfall patterns including the frequency and intensity of rainfall events. Although there may be isolated ponds forming within depressions on the floor of the workings, these areas will flood over time and decant towards the following down-slope depression or adit entrance.

The groundwater flow patterns within such workings may be described as active flow paths between stagnant pools of relatively poor quality water. The degree of mixing within the pools will depend on site specific conditions such as the depth of the pool, etc. The ability to flood large portions of the mine will generally depend on the

floor contours of the mine, as well as the integrity of the boundary pillars along the edges of the mine and their ability to hold back water.

These types of mines always decant unless some form of remedial measures are undertaken to effectively block the mine entrance to flood the workings. Surface decant volumes are likely to be in a similar order of magnitude to that during the active mining unless remedial measures are implemented to reduce the recharge to the workings. The time taken to decant will depend on the water levels within the isolated ponds at the time of mine closure. Any remaining storage capacity within isolated ponds will delay the time to decant accordingly.

From a geochemical perspective, the underground adit coal mine (see Figure C8 above) could contain the following features that have an impact on long term water quality:

Total extraction mining:

In this scenario, the bulk of the coal, containing sulphide minerals, will have been removed and the sulphide minerals that could cause water quality deterioration would be a small amount of coal and any sulphide minerals in the roof material that has collapsed into the workings. Depending on the base minerals contained in the roof material, there may be a significant amount of neutralisation occurring in these areas, with the resultant production of secondary minerals. These secondary minerals will be a potential store of contaminants to be released over the long term. The total extraction scenario will also present the water with a high surface area of material that will lead to increased acid and base reactions occurring.

Split and quartering:

In this scenario, where the pillars have been reduced in size to the point where the roof has collapsed, significant amounts of sulphide containing coal could still be present, leading to a potentially more acid generating system than for total extraction mining. In other respects, this system will have similar features to the total extraction scenario, including production of secondary minerals and high surface areas. At this stage, from a conceptual viewpoint, this scenario is seen as the worst case from a water quality deterioration viewpoint.

Bord and pillar mining:

In this scenario, where the roof has not collapsed, the water is deemed to be primarily in contact with sulphide

minerals, although base minerals on the floor may also play a role. The primary feature of this scenario, however, is that the surface area of minerals in contact with the water is relatively small, although fines and particles dropping from the pillars and roof due to weathering and stress will contribute to an increase in reactive surface area.

Voids containing slurry:

In certain cases coal beneficiation plants may pump slurry from the plant into underground compartments. The slurry tends to be fairly impermeable, resulting in a reduction in the amount of kinetically driven aerobic oxidation processes, particularly if the slurry retains a high water content. On the other hand, however, due to the small particle size and high particle surface area, base minerals that can react in anaerobic conditions, may have a much higher availability and may dominate geochemically. This type of mine area may also accumulate large amounts of secondary minerals that could be released in future when the base potential is exhausted or incapable of dealing with the incoming acid load.

Voids containing liming plant sludge:

Certain mines that operate liming plants have historically employed the practice of discharging liming plant sludge into underground voids. While this sludge will have some inherent residual neutralising capability, the sludge also contains large amounts of heavy metal precipitates (mainly as hydroxides) that will be released and redissolved when the sludge's neutralising capacity has been exhausted. These areas are considered to pose a very high geochemical and water quality risk.

Flooding of mine workings:

Wherever the workings are flooded, the geochemical processes will undergo a significant change. The sulphide oxidation process will essentially stop and the primary processes affecting water quality will be dissolution of stored secondary minerals and dissolution of available base minerals. The controlling process will tend to move away from kinetic to equilibrium driven processes. Flushing and dilution processes will also need to be taken into account. Complete flooding is, therefore, advantageous from a geochemical and water quality viewpoint. If the degree of flooding changes, i.e. water levels regularly fluctuate, then the situation is different and kinetic sulphide oxidation processes may dominate. This situation could, due to the increased surface area of material brought into contact with the

water, be geochemically worse than a totally unflooded and fully aerobic system.

The differences between the conceptual geochemical models for mined out seams are depicted in Figure C9.

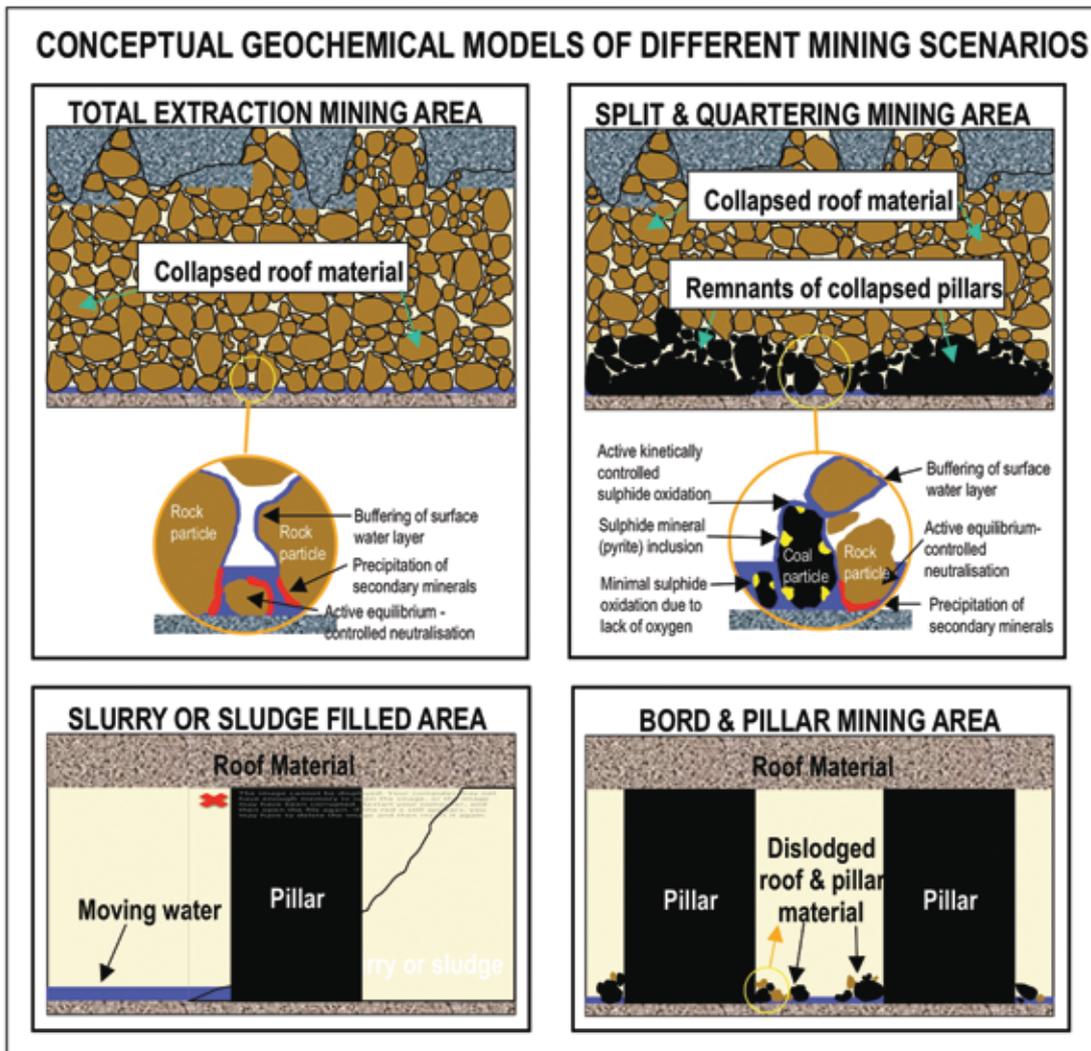
Deep Witwatersrand type gold mines

According to Scott (1995), gold mining began on the Witwatersrand soon after the discovery of gold in 1886. Preliminary mining occurred along the outcrop in the Central Rand. Shafts were required to access the gold reefs in other areas since the sediments were covered by Transvaal and Karoo Strata.

The quantity of water ingress into the workings is dependant on the origin of the water. There are four broad categories of water ingress as follows:

- Connate water trapped within the sediments at the time of deposition
- Natural groundwater recharge
- Additional water ingress along the outcrop workings from surface runoff and mine infrastructure
- Water ingress from overlying dolomitic aquifers

Figure C9: Conceptual geochemical models for different scenarios in the mined out coal seams



According to the work undertaken by Van Biljon (1995) at Joel Mine in the Free State Goldfields, the majority of water ingress in those workings originates from connate water that is located along structural features such as faults. Recharge from the overlying Karoo sediments are considered to be negligible in these areas. Groundwater ingress into the workings is therefore managed by means of a dewatering program that includes cover drilling ahead of mine development. Once the sediments have been dewatered, the cover holes remain essentially dry. It is anticipated that groundwater recovery under these circumstances would not be complete since there is a limited source of connate water that will only be marginally replenished by recharge from the overlying strata. Complete flooding of the workings upon mine closure may not occur or will be at a slow rate depending on the extent of the dewatering that occurred during the mining phase.

Natural groundwater recharge in the order of 6.5% of the mean annual precipitation takes place within the Witwatersrand strata in the West and Central Rand (Krantz, 1999). However, there is a shallow aquifer located between topographic surface and a depth of some 60m. It is thought that rainfall recharge occurs primarily to this aquifer. As the groundwater flows laterally within this aquifer, it is intersected by existing mine workings and geological structures. These features act as preferential pathways for groundwater recharge into the deeper underground workings. Although the water ingress is influenced by rainfall patterns, there is a lag time of some 2-4 months between the rainfall events and the increase in the water ingress to the mine workings.

The rate of groundwater level recovery upon mine closure is dependant on the storage capacity of the mine voids and the water ingress over time. Partial closure of the mine workings should be compensated for particularly where the dip of the reef is relatively flat ($<15^\circ$). It is anticipated that the mines will generally decant on surface at the lowest shaft or adit where the water ingress is dependant on rainfall and surface water runoff. The rate of water level recovery will be accelerated where additional water ingress originates from opencast pits and surface water runoff along outcrop workings.

There is still controversy surrounding the degree of mixing that is associated with active recharge in gold mine workings. However, profiling work in the West Rand has indicated the presence of a relatively fresh water lens close to surface of the current water level elevation within the workings. The decant volume will not necessarily be

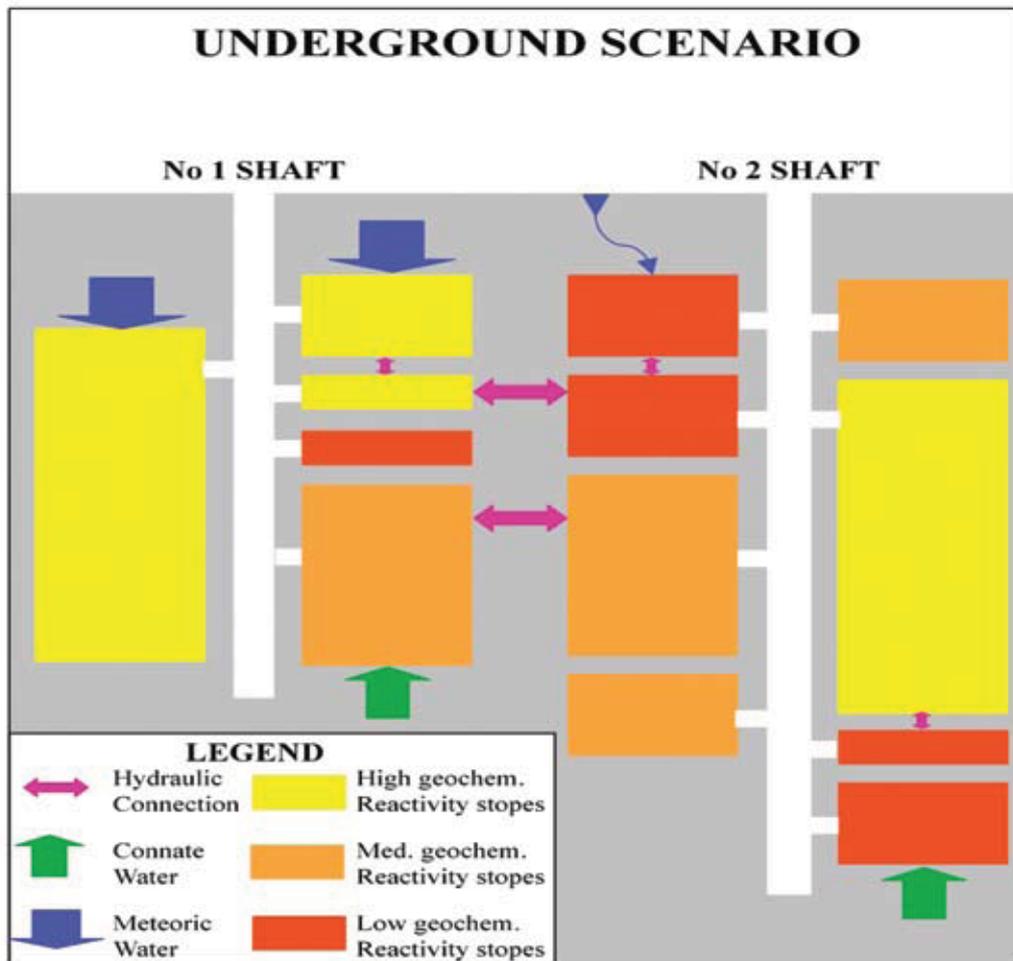
the same as the water ingress to the workings during active mining since recharge to the mine workings may decrease in places as the ambient groundwater gradients are re-established.

Water ingress from overlying dolomitic aquifers is common in the Far Western Basin, along the West Wits line. The overlying dolomitic compartments have often been dewatered and under these circumstances, water ingress into the workings generally occurs along geological features such as faults and dykes. The piezometric head within the dolomites has acted as the primary driving force for the groundwater into the mine workings. It has been observed that the water ingress into the workings has decreased as the groundwater level within the overlying dolomitic groundwater compartment has been lowered over time. Any water originating from storage within the dolomites has therefore been removed. However, it has been anticipated that the groundwater ingress into the workings will stabilize at the recharge rate to the dolomitic compartment as a minimum volume.

Once again, the rate of the flooding of the mine workings will be dependant on the recharge to the dolomitic compartments under these circumstances. The mine voids at the time of mine closure will also influence the rate of groundwater level recovery. Mine shafts have often been sealed within the dolomites to reduce the potential for future poor quality mine water to seep into the dolomites upon recovery. However, there is still a debate as to whether there would be sufficient mixing between the active recharge water to the dolomites and the mine water to generate poorer quality water that will result in the deterioration of the overall water quality within the dolomites.

A notable exception is the West Rand where it has been estimated by Krantz (1999) that some 20% of the water ingress into the workings originates from the dolomites. A groundwater reversal will take place as the water level recovers past the dolomites located at an elevation of 1540mamsl on the way to topographic surface on the Witwatersrand Ridge located at 1729mamsl. There is the potential for poor quality mine water to seep into the neighbouring dolomites under these circumstances. The amount of water seeping into the dolomites is dependent on the groundwater gradient towards the dolomites and the permeability of the geological structures along which the groundwater flow would primarily take place.

Figure C10: Conceptual Underground Model



From a combined geochemical and geohydrological perspective, a simplified conceptual underground scenario can be presented as shown in Figure C10 above. In this figure, the impact of hydraulic connections, combined with the geochemical alteration possibilities along specified flowpaths is evident. Some areas in specific shafts could be hydrologically isolated with comparative ease. When the geochemical characteristics of such areas are considered, discrete areas with varying potential to give rise to poor quality water may be identified and managed accordingly (see Section C4.1 above).

Deep bord and pillar coal mining

Deep bord and pillar coal mining may be defined as mining which occurs at such a depth so as to not disturb the overlying aquifers. According to Hodgson and Krantz (1998) water within deep coal mines in the Olifants

Catchment is scarce. This may be attributed in part to the considerably lower vertical permeability compared to the horizontal permeability of the Karoo sediments. Water ingress into the workings occurs primarily along geological structures such as faults and dykes. However, these mine workings may generally be described as dry. Since water ingress does occur primarily from the overlying sediments, it is expected that these workings will ultimately flood. However, given that the recharge is relatively low, this process is expected to take a considerable period of time.

Shallow bord and pillar coal mining

Shallow bord and pillar mining may be defined as mining that occurs at a depth where the base of the weathering profile is intersected. These mines are generally older and are usually defunct. Subsidence is

a common phenomenon in these areas since the roof material consists of weathered Karoo strata. These collapsed structures may result in a checker board effect on surface. Rainfall recharge and ingress from surface runoff is a lot higher under these circumstances since the subsidence sinkholes promote water ingress into the mine. Subsidence may also be accelerated as the groundwater level recovers in the mine workings since this may weaken the pillars (Younger and Adams,1999).

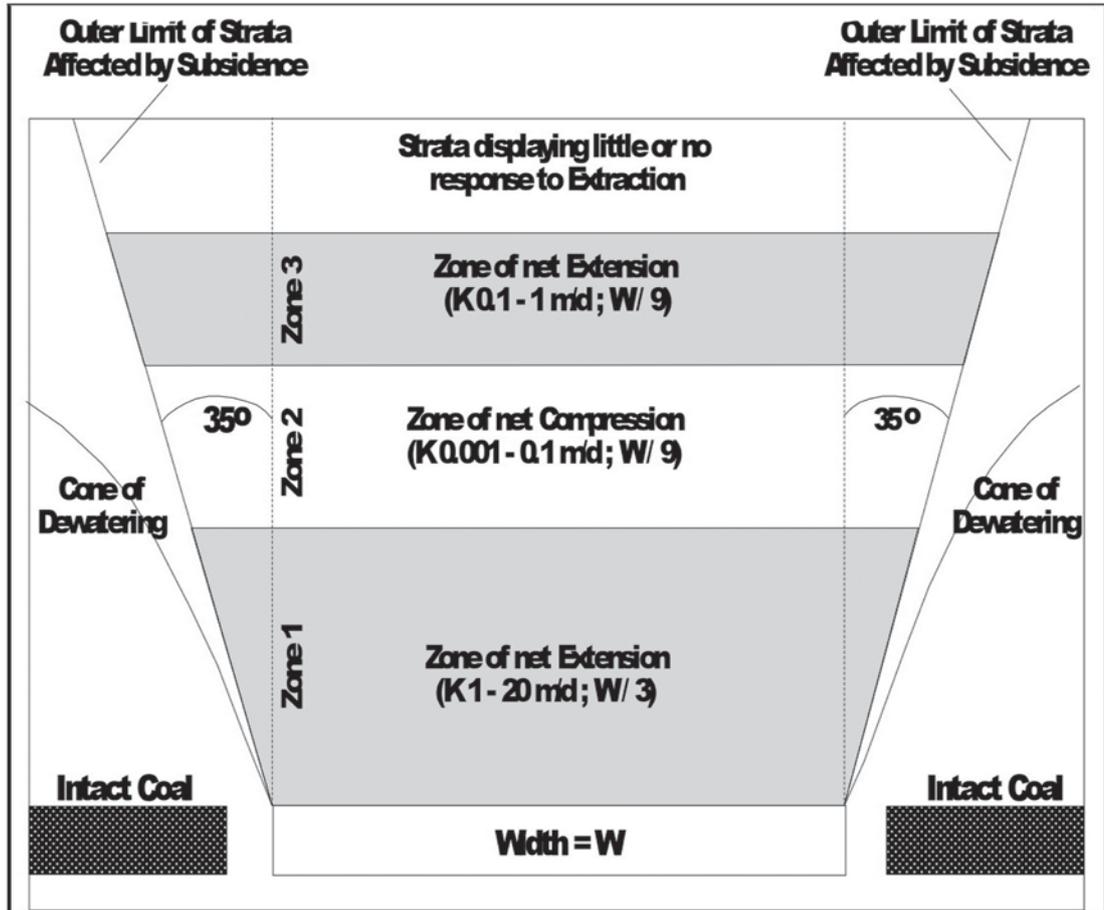
These workings will probably decant if they are not connected to deeper workings since they are relatively close to surface. The amount of flooding is primarily dependant on the coal floor contours and whether the decant point is free flowing or partially blocked due to backfill or subsidence. Decant volumes will probably be similar to that of the active pumping volumes since the majority of the recharge originates from rainfall and surface runoff.

High extraction coal mining

High extraction coal mining occurs where the coal is completely removed while allowing the roof to collapse or goaf. The extent of the goafing is dependant on the depth of the mining, the height of the coal seam and the width of the high extraction panel. The zones of extension and compression as a result of goafing above high extraction panels are shown graphically in Figure C11. During goafing, subvertical cracks are formed at the edge of the panels which may extend to surface in certain instances. The integrity of the overlying strata is compromised as a result.

Hodgson and Krantz (1998) have observed that the highest ingress of water occurs immediately after the goafing of the roof due to the dewatering of the overlying aquifers. This volume of water decreases to a minimum volume which is representative of the rainfall recharge and the lateral groundwater flow towards the dewatering cone associated with the panel. These panels therefore act as important sources of water ingress into the underground workings. Such panels are often connected to other workings. The groundwater recovery would therefore depend on the coal floor contours, the decant elevation and the mine void remaining in the workings. Unlike bord and pillar mining where the mine void is intact, the mining void in high extraction panels is distributed through the roof strata as the overlying strata collapses. This has important implications for the prediction of the mine void space with depth to determine the anticipated groundwater recovery upon closure.

Figure C11: Example of the Zones of Compression and or Extension above high extraction panels (after Younger and Adams, 1999)



**APPENDIX D
EXAMPLE OF
CONCEPTUAL
MODEL REPORT
FOR WASTE
DISPOSAL FACILITY**

D1. INTRODUCTION

In order to investigate the closure options for the residue disposal facility, consisting of the slimes dam and coarse waste dump at the site, a conceptual model will be developed. The geohydrological and geochemical modelling approaches will be based on this conceptual model.

Ore is received by truck from the mine. The ore arrives in crushed aggregate form and undergoes further crushing in ball mills. This produces a fine slurry from which the product can be separated from other impurities by means of magnetic separation and a roasting process. Two waste products are produced, namely a relatively coarse waste and a very fine slime in solution. The coarse waste material is conveyed to the residue disposal facility (RDF) and the slimes are pumped up to the same RDF and deposited in a central void created within the RDF. Excess water is pumped back to the plant for re-use.

Cut-off trenches have been constructed to intercept the surface seepage, but infiltration into the groundwater continues, assisted by the fact that the RDF is located above an undermined area - the area under the RDF is extensively and possibly entirely undermined. There is at least one known sinkhole located underneath the dump that extends from surface to the underground workings. Video footage obtained from a borehole adjacent to the RDF also suggests that the mine void has collapsed, probably creating additional seepage flowpaths from the RDF to the mine workings.

The current waste disposal system has been in place since 1957. Over the time span of the RDF, considerable changes have evolved in regard to the environmental management of the site. The plant has implemented measures to contain surface water on the residue facility and to return this water for use in the plant. The coarse waste material is produced at a rate of 65 000 tons per month and will rise to an estimated 89 000 tons per month.

D2. CHARACTERISTICS OF THE RESIDUE DISPOSAL FACILITY

The RDF consists of different materials, which have more than one name given to them by various institutions. We must therefore define the following terms for use in this project:

- **Residue disposal facility (RDF)** refers to the whole dump, including the coarse waste dump material and the slimes dam.
- **Coarse waste dump** refers to the “coarse” waste material produced by the plant and sometimes referred to by them as tailings. It also contains small amounts of fly ash waste from the burning of coal in various processes.
- **Slimes** refers to the “fine” waste material deposited as a slurry into a dam (**Slimes dam**) located within the coarse waste dump. The slimes materials largely consists of the gangue and dewatered material pumped from the milling section of the plant, but also contains other waste water and solids pumped from a pollution control dam and other process discharges. It is assumed that the physical properties of the slimes dam are fairly similar to the properties of a gold and /or uranium tailings dam.

Although the coarse waste dump material is classified as coarse when compared to the slimes material, the coarse waste dump material consists of relatively fine grained material. Grain size analysis conducted by Weber Zenon and Associates (2002) indicated that 90 % of all particles are smaller than 2 mm in diameter.

The slimes and the coarse waste dump material have similar geotechnical characteristics. Both materials classify as a silty sand or a sandy silt mixture, however, the slimes material (96% <0.427mm) is much finer than the coarse waste materials (90% <2mm).

During a site visit, pieces of conveyer belt were observed on the coarse waste dump material. These large objects will assist in the formation of preferential pathways in the coarse waste dump material, however, it is uncertain if preferential pathways will form in the slimes dam. The slimes are assumed to be wet and very fine and will therefore probably fill any hydraulic conduit that may exist.

Fine waste deposits exhibit the following characteristics (DWAF, 2002):

- Particles are relatively uniform in size.
- The water table is typically elevated with a large portion of the facility in a saturated state.
- The relative proportion of seepage emanating from the toe of the dump relative to that seeping into the underlying aquifer is dependent on the permeability contrast between the waste material and the underlying geology and the difference between horizontal and vertical permeability of the coarse waste dump material.

The driving forces and processes that need to be explicitly considered when assessing the quality of water draining from the RDF are listed below:

- Rainfall onto the top and sides of the RDF.
- Runoff from the sides of the coarse waste dump.
- Evaporation and evapotranspiration from the top and sides of the RDF.
- Infiltration into the dump along preferential flow paths towards the phreatic surface (water table).
- Diffusion of oxygen into the RDF.
- Seepage of contaminated water from the RDF (either at the toe or through the base).

The evaporation from the RDF is not measured and is therefore an uncertain parameter. Evaporation from the slimes dam will be different than evaporation from the coarse waste dump. Potential evaporation depends on atmospheric conditions, but the actual flux across the upper boundary is limited by the ability of the material to transmit water from below. In fine active tailings, evaporation effects are large and this causes a drying of the upper surface. This drying of the upper surface, in turn, causes an upward flux from the water table.

Runoff is another uncertain parameter. The perimeter of the RDF is raised, prohibiting any runoff from the top of the RDF, therefore runoff can only occur from the sides of the RDF. A limited amount of grass occurs on the slopes, which may reduce runoff in the areas where it does occur. However, most of the RDF does not have any cover on the slopes. The International Atomic Energy Agency (1990) gives a runoff value of 19 % for uranium tailings with no cover. However, tailings dams have relatively well compacted, hard slopes with strong evidence of erosion, while the RDF has soft unconsolidated slopes and is therefore assumed to have higher infiltration and less runoff. Special experiments were undertaken to establish the runoff, infiltration and evaporation parameters for coarse material as part of this project.

D2.1 SLIMES

A separate study compared the grading analysis of slimes in 2002 with the grading analysis of 1998 and observed significant differences in fines. This has been attributed to a change from four small ball mills to two large ball mills that were installed in 1999. However, the grading analysis conducted in April 2002 shows a lower percentage of fines (0.075 mm) than that of the work conducted in both 1998 and 2002. Table D1 gives the grading analysis for the years 1998 and 2002 as well as the grading analysis from the second 2002 study.

Table D1: Comparison of slimes grading from different measurements.

Sieve size (mm)	1998 measurements SLIMES	2002 measurements SLIMES	2002 measurements Second Study SLIMES	2002 measurements COARSE WASTE
4.750	100	100	100	96
0.425	99.7	94	96	78
0.15	67.7	72	40	42
0.075	54.1	60	23	19
0.050			17	15
0.043	7.6	45		
0.005			3	5

The permeability of the slimes is assumed to be very low due to the fine and well sorted nature of the particles. However, a wide range of permeability values is found in the literature. Company A reports a permeability of 8×10^{-8} m/second (6.9×10^{-5} m/day) for the slimes, while Company B measured values of between

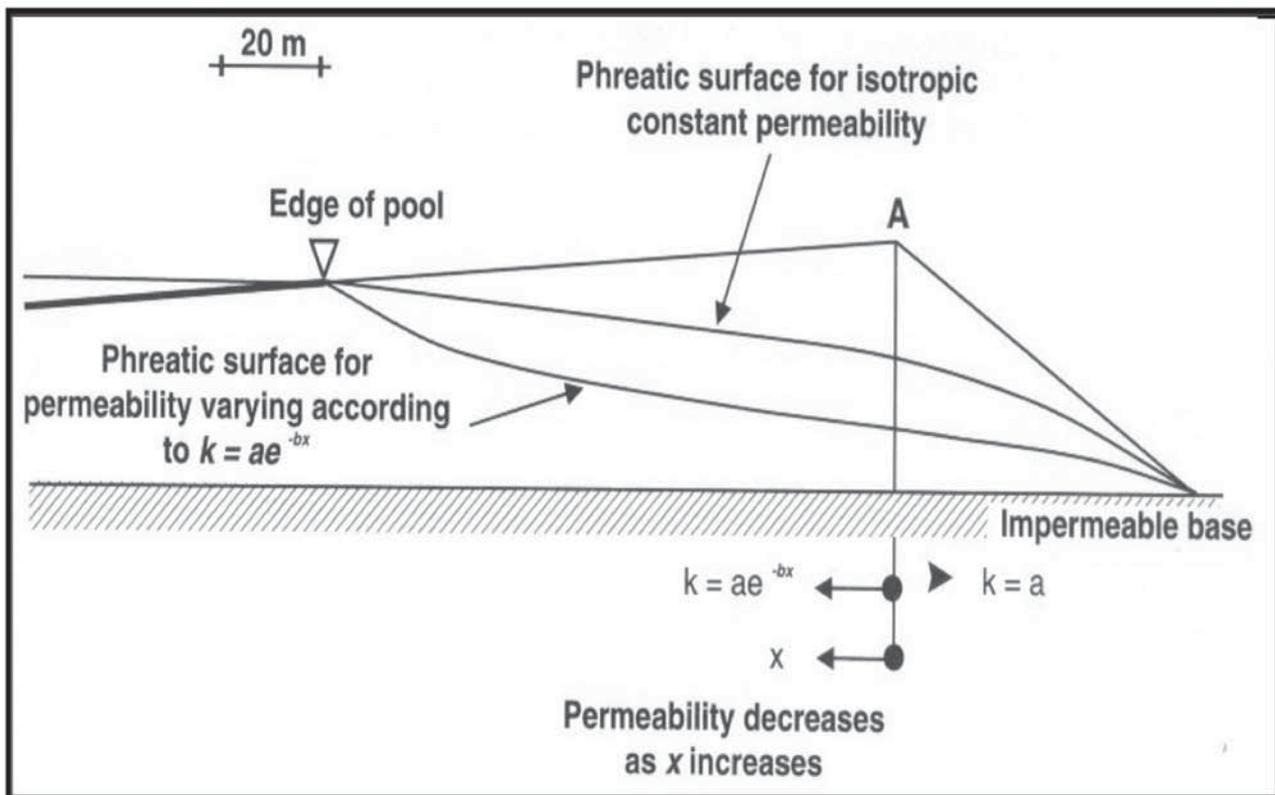
1.2×10^{-3} cm/second (1.04 m/day) and 2.7×10^{-3} cm/second (2.33 m/day) for slimes. These values reported by Company B seem unrealistically high for fine grained slimes and is within the same order of magnitude as was measured for the coarse waste material. In terms of reported particle size, the slimes sample does not differ much from the coarse sample. It is therefore assumed that the sample was taken from the upper beach area of the slimes dam where the coarser particles will accumulate. Company C (1992) reports a permeability of 9.4×10^{-4} cm/second (0.8 m/day) for the beach area of the slimes dam. This value is lower than the value of Company B (2002). Due to the large range in reported values, field permeability measurements were taken as part of this impact prediction project.

To summarise, it must be noted that the grading of the material is relative to where the sample was taken (closer to the middle of the slimes dam or near the perimeter of the beach area) and also relative to when the sample was taken (due to the nature of the

slimes, containing storm water runoff, samples taken shortly after a high intensity rainfall event, may have a very different analysis to a sample taken at some other time).

Active pool and beach areas exist on the slimes dam. Slimes are deposited near the edge of the dam and flow towards the middle of the dam. Due to gravitational sorting a gradient of decreasing permeability exists from the outside to the inside of the tailings dam (Blight, 1987). As a result of this variation in permeability, the phreatic surface of the dam will be depressed (Figure D1). Observations during a site visit suggest that approximately 75% is pool and the balance is apportioned between wet and dry beach. It was also observed at the time of the site visit that more than half of the pool was lapping against the coarse waste dump and not onto the beach resulting in much higher infiltration than would otherwise be the case (Figure D2).

Figure D1: Effect of varying permeability on position of phreatic surface in a slimes dam (Chamber of Mines. 1996)



The presence of finer material such as clay within the slimes dam is not confirmed. Should these fine materials be present in the slimes dam, it would contribute to a more layered system that will inhibit vertical flow. Company B (2002) make the assumption that seepage flow within the slimes dam is predominantly downward fairly uninhibited, based on piezometric measurements that shows depressed water levels. The depressed water level is expected due to the gradient of decreasing permeability as described above. For the purpose of this study, additional piezometers were installed in order to establish water level profiles.

Figure D2: Pool area of slimes dam showing no beach (top) and a large beach area (bottom)



According to Company A (2002), the slimes would contribute 5 % by volume of total slurry water as seepage that would migrate into the mine workings, while 18 % would be as surface seepage.

Company D (2002b) calculated a recharge value of 10 to 12 % of rainfall for the beach areas of the slimes dam, while they used the saturated hydraulic conductivity (1×10^{-6} m/s or 0.086 m/day) for the pool areas.

D2.2 COARSE WASTE DUMP

The particle sizes of coarse waste dump material were found to be fine, but less well sorted than the slimes particles. The percentage gravel in the coarse waste dump material was measured to be 10.3%. Sieve analysis showed that 96 % of the particles are smaller than 0.425 mm in diameter, 42 % are smaller than 0.15 mm in diameter and 6 % of the particles are smaller than 0.005 mm in diameter.

The permeability of coarse waste dump material as given by Company B (2002) as 1.3×10^{-3} cm/second (1.123 m/day). Company D (2002b) calculated a recharge value of 10 to 12 % of rainfall for the coarse waste dump. This value appears low given no runoff from the top. An experiment was conducted as part of this project to determine the infiltration into coarse waste dump material.

Layering was observed on the side of the RDF (Figure D3). This is similar to what is found in gold tailings dams and would probably cause horizontal hydraulic conductivity to be larger than vertical hydraulic conductivity. Due to the nature of natural deposits, an anisotropy factor of 10 is generally assumed, meaning horizontal hydraulic conductivity is one order of magnitude (10 times) larger than vertical hydraulic conductivity. In the RDF the anisotropy factor may be higher than 10.

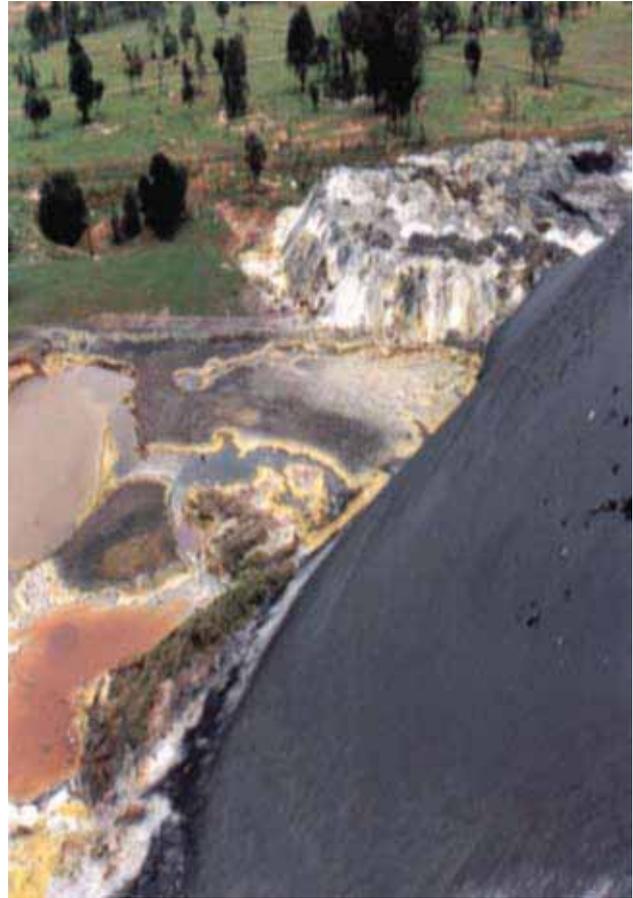
Precipitates with greenish, brownish, and/or yellowish colour were frequently seen in the seepage surrounding the RDF

(Figure D4). Seepage emanating from the toe of the dump is quite extensive (Figure D4), indicating that a significant volume of seepage is not reporting to the underground workings beneath the RDF. In a situation where a waste dump such as the RDF has a much higher permeability than the underlying soil, it can be expected that a significant portion of the seepage will report as toe seepage rather than infiltrating into the underlying soil. This differential in permeability is confirmed by the borehole logs that indicate a clay layer beneath the RDF.

Figure D3: Photo of the RDF side wall showing a layering structure within the dump



Figure D4: Seepage from the toe of the dump



D2.3 UNDERLYING GEOLOGY

D2.3.1 Regional Geology

The area is underlain by rocks from the Eccra and Dwyka Formations of the Karoo Sequence. The lithology of the Eccra Formation comprises of shale, shaly sandstone, grit, sandstone conglomerate and coal in places, while tillite predominates in the Dwyka Formation. Dolerite intrusions in the form of dykes and sills could also occur. (Company C, 1992).

D2.3.2 Local Geology

Company C (1992) observed a well developed clay layer occurring over the entire plant area. The thickness of the clay layer ranges from 6 to 14 m with the thicker areas in the vicinity of the RDF. Beneath the clay layer, the sequence comprises of a medium to coarse grained carbonaceous shale, a gritty sandstone and a coal seam. No dolerite intrusions were intersected during the drilling of 10 boreholes in the plant area. The entire area around the residue disposal facility is undermined and collapse to surface has occurred widely (Company E, 2001). Plans obtained

from Company D (2002a) indicate that the area directly under the RDF is more than 90% undermined.

The natural hydraulic conductivity of the shale is low, but mining activities enhanced it. The hydraulic conductivity of Karoo aquifers ranges between 0.00086 m/day (Company D, 2002b) and 0.04 m/day (Krantz, 2002).

The following assumptions are made for the geology under the RDF:

- The clay layer occurs under the entire RDF.
- The entire area under the RDF has been undermined.
- No dolerite intrusions occur under the RDF.
- There are subsidence features between the underground workings and the RDF, resulting in the establishment of hydraulic conduits.

In general, clays have a high sorption coefficient, which means that contaminants will adsorb to the clay particles. This may reduce the pollutant levels in water. Permeability values for the clay layer are not given, but Company C (1992) suggest that the vertical permeability will be almost insignificant. Todd (1980) reports that hydraulic conductivity for clay is 2×10^{-4} m/day while Anderson and Woessner (1992) give a range of hydraulic conductivity values for clay between 5×10^{-4} and 5×10^{-7} m/day.

The water level is reported to be between 2.5 and 5 m below surface following surface topography (Gardner et al, 2001). Direction of flow is south and west, draining towards the river. Groundwater yield is insignificant with the exception of borehole B3 that yields 4068 l/hr.

D3. CONCEPTUAL MODEL

The conceptual model is constructed with a series of nodes with different properties that will be used in the hydrological and geochemical modelling. The following characteristics were considered in choosing nodes for the conceptual model:

- Material types;
- Residue deposits disposal options;
- Saturation characteristics; and
- Interconnectivity between the RDF and underground coal mine workings.

The conceptual model has also been constructed to enable the consideration of the effects of removing the coarse dump while leaving the slimes dam intact.

The materials identified in the RDF are slimes and coarse waste dump material.

It is understood that the sale of the coarse dump is under discussion. In this case, the major part of the coarse dump will be removed, but a certain amount of the coarse dump must be left to retain the slimes dam. The conceptual model must therefore be constructed to enable the sale option to be evaluated.

Sinkholes and surface subsidence due to underground mine workings occur in all the areas surrounding the RDF. It is therefore assumed that there are sinkholes under the dump causing the coarse waste dump material and the underlying soil to move to the floor of the underground workings. This will be added as a node in the geochemical model.

O₂ will be diffused in the unsaturated zone and limited in the saturation zone. Further nodes need to be considered to distinguish between saturated and unsaturated zones.

Figure D5 shows a plan view of the RDF in terms of its geometric features. A cross section (A-A') goes through the RDF and shows the section features of the RDF (Figure D6).

Figure D5: Plan view of the RDF, drawn from map supplied by Company A (2000). Cross section A-A' is diagrammatically illustrated in Figure D6

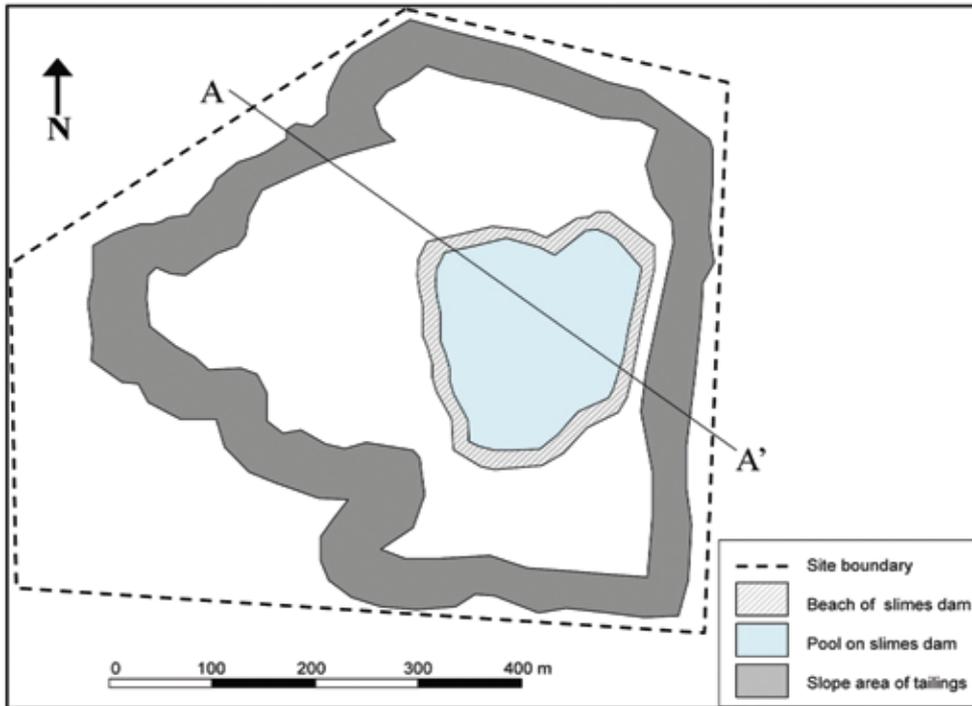
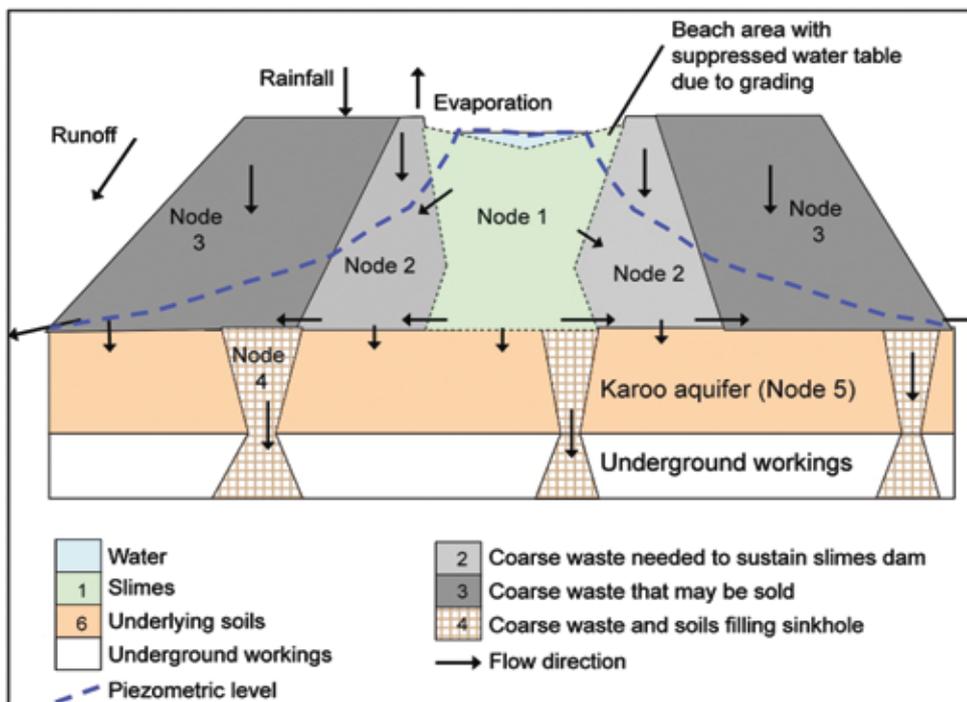


Figure D6: Diagrammatical illustration of the processes and nodes considered in the conceptual model



Based on the abovementioned considerations for the development of a conceptual model, 5 nodes have been identified (Figure D6). The processes and nodes that will be considered in the conceptual model, are illustrated in Figure D6.

The processes that need to be considered for the water balance are:

- Rainfall
- Additional water added with slimes (minus water pumped back to plant)
- Surface moisture on coarse material
- Interstitial water retained within the slimes
- Evaporation
- Runoff
- Infiltration
- Seepage

The nodes to be considered are:

- Node 1: Slimes
- Node 2: Coarse waste dump material that is needed to support the slimes dam should some of the coarse waste dump material be removed on sale thereof. This node is subdivided into node 2A, the unsaturated part of node 2, and node 2B, which is the saturated part of node 2.
- Node 3: Coarse waste dump material that can be removed / sold
- Node 4: Coarse waste dump material and soil that moved downward due to the formation of pipes and sinkholes. The amount of pipes and sinkholes are unknown and therefore the amount of seepage into the underground workings is unknown. It is suggested that a borehole be drilled into the underground workings at a point that is most likely to contain seepage. Using a camera lowered into this borehole, a qualitative observation can be made to confirm the presence of seepage.
- Node 5: The underlying soils and mixing zone for all the nodes

Note that due to extensive toe seepage that is evident, not all seepage can report to node 5, which has been reported to have a very low infiltration. It is assumed that the dominant flowpath is horizontally through node 3 and out of the toe of the dump, some flow will be vertically through node 4 directly to the underground workings and that very little flow will infiltrate into node 5

As this modelling exercise is aimed at establishing pollution profiles at dam decommissioning and after closure, we define the **base case scenario** to be the decommissioned RDF after the last load of waste material has been deposited thereon and

before any rehabilitation measures have been applied. At this point, the slimes dam will still be fully saturated and the coarse dump will still retain a similar shape as it does at present, although it will be somewhat larger. For the purpose of modelling, the final volume of material within the RDF will be as follows:

- **Coarse waste:** Present volume + (A months x B tons/month) + (C months x D tons/month)
- **Slimes:** Present volume + (E months x F tons/month) + (G months x H tons/month)

Where: A, B, E and F are values that apply for the one operation and
C, D, G and H are values that apply for the second operation.

The observed slimes disposal and pool management practice that causes the slimes dam not to have a beach area on all sides of the slimes dam is not acceptable. It is of utmost importance that corrective measures are taken in order to create and maintain a beach all round the slimes dam. It is believed that when slimes are properly deposited through all the nozzles available around the dam, that the beach area will be created automatically. It is assumed that this practice will be corrected and that the beach areas will be established all around the slimes dam at the time of closure.

It is further assumed that the slimes solution will be allowed to settle out and that all remaining water will be removed for closure of the RDF. This should not cause any management problems, because the plant will still be operating at closure of the RDF.

D3.1 NODE 1

Node 1 consists of the slimes dam. In the base case, it is assumed to be fully saturated with water.

D3.1.1 Hydrology assumptions for Node 1

The following assumptions are made for Node 1:

- Runoff is assumed as zero for Node 1.
- An experiment will be conducted to determine the infiltration into coarse waste dump material.
- It is assumed that this practice will be corrected and that the beach areas will be established all around the slimes dam at the time of closure.
- The saturated hydraulic conductivity will be measured by the consultant.

D3.1.2 Geochemical assumptions for Node 1

The following assumptions are made for geochemical modelling:

- Fully saturated with water;
- The slimes contain different kinds of waste materials and any sampling programme must take this into account. Extensive sampling and analyses of the slimes will be carried out to obtain mineralogical components, size fraction, porosity and permeability of the slimes.
- Extremely low permeability ($8e^{-10}$ m/s, Company A, October 2002). Field measurements will be undertaken in this study to provide first hand information for permeability;
- Use slimes water quality as start pore water quality;
- The recharge water is rainfall whose water quality needs to be obtained from analysis of 2 water samples collected by the plant;
- Infiltration data needs to be obtained from water balance from section D3.1.1.
- Moisture contents from the setting tests for one sample prepared at densities of 1.05 and 1.36t/m³ reported by Company A (October 2002): At 1.05 t/m³ 20.9% (drained) and 63.1% (undrained test); and at 1.36 t/m³, 39.4% (drained test), 59.5% (undrained test). The first hand information for moisture content will be obtained from the analyses of samples in this study.
- Mineralogical data (Company A, 2002): ilmenite (FeTiO₃), goethite (FeO(OH)), albite/calcian ((Na, Ca)Al(Si,Al)₃O₈), Hematite/Magnetite (Fe₂O₃/Fe₃O₄), kaolinite (nacite/halloysite, Al₂Si₂O₅(OH)₄), chlorite (clinochlore, (Mg,Fe)₆(Si,Al)₄O₁₀(OH)₈), mica/altered mica (annite, Mg-Fe-K-Al-silicate), and quartz. XRD analysis of samples collected in this study will provide quantitative mineral components. Microscopic study will assist further detailed study on minerals containing particular contaminants of concern, e.g. hematite/magnetite).
- Slurry water chemistry provided by Company A (2002) indicates very high TDS, Na, and SO₄ and high B, Mn and Na concentrations. Slime water quality will be analysed and will be used as initial water quality data for geochemical modelling.

D3.2 NODE 2

Node 2 comprises coarse waste dump material that is needed to sustain the slimes dam should some of the coarse waste dump materials be sold and removed.

D3.2.1 Hydrology assumptions for Node 2A

The following assumptions are made for Node 2A:

- Runoff is assumed as zero for Node 2A, however for a model after removal of the bulk of the coarse waste dump material, runoff will be considered;
- The saturated hydraulic conductivity will be measured by the consultant;
- An experiment will be conducted to determine the infiltration into coarse waste dump material.

D3.2.2 Geochemical assumptions for Node 2A

The following assumptions are made for Node 2A:

- O₂ diffusion in this zone will play an important role for oxidation of magnetite which contains particular contaminants of concern. The O₂ partial pressure will be calculated from site measurements to be taken by the consultant;
- Water infiltration refers to Section D3.2.1;
- Existence of larger grain sizes of the materials which affect mineral surface area calculation;
- Smaller mineral surface areas for different minerals than in the slimes;
- Materials are enriched in magnetite, feldspar, clays, Fe-hydroxides, Fe-oxides;
- Adsorption capability in this node is important;
- Water extraction data multi-elements ICPMS Scan will be used as initial pore water quality data because no slime water will recharge to this node.
- Rainfall water as the only recharge water will be analysed for water quality for input data for the geochemical modelling; and
- Evaporation rate of rainfall to be taken from consultant's experiments and will be considered for geochemical modelling.

D3.2.3 Hydrology assumptions for Node 2B

The following assumptions are made for Node 2B:

- Runoff is assumed as zero for Node 2B, however for a model after removal of the bulk of the coarse waste dump material, runoff will be considered;
- Infiltration into Node 2B derives from Nodes 1 and 2A;
- The saturated hydraulic conductivity will be measured by the consultant;
- An experiment will be conducted to determine the infiltration into coarse waste dump material.

D3.2.4 Geochemical assumptions for Node 2B

The following assumptions are made for Node 2B:

- No O₂ diffusion in this zone because it is fully saturated with water.
- Initial water will be the slime water.
- All the others will be the same as Node 2A.

D3.3 NODE 3

Node 3 is coarse waste dump material that is far enough away from the slimes to enable it to be sold and removed without influencing the stability of the slimes dam.

D3.3.1 Hydrology assumptions for Node 3

The following assumptions are made for Node 3:

- Runoff is assumed to be lower than 19 % (International Atomic Energy Agency, 1990) for Node 3.
- The saturated hydraulic conductivity will be measured by the consultant;
- An experiment will be conducted to determine the infiltration and runoff into coarse waste dump material.

D3.3.2 Geochemical assumptions for Node 3

It is the same as for node 2A, but infiltration of rainwater is less.

D3.4 NODE 4

Node 4 consists of coarse waste dump material and soil that moved downward due to the formation of a sinkhole. Details for this node need to be investigated further. It is suggested that a borehole be drilled into the underground workings at a point that is most likely to contain seepage. Using a camera lowered into this borehole, a qualitative observation can be made to confirm the presence of seepage.

D3.4.1 Hydrology assumptions for Node 4

Permeability is assumed to be higher than 1.123 m/day (Company B, 2002) but will be measured on site.

D3.4.2 Geochemical assumptions for Node 4

The following assumptions are made for Node 4:

- Fully saturated with water; and
- Adsorption by clays plays an important role in the geochemical process.

D3.5 NODE 5

Node 5 is the geological formation underlying the dump.

D3.5.1 Hydrology assumptions for Node 5

It is assumed that the clay layer plays a major role in restricting downward flow. The recharge into this layer is assumed to be around 2.5 % of rainfall (Department of Water Affairs and Forestry, 1986).

D3.5.2 Geochemical assumptions for Node 5

A thick soil / clay layer exists underneath the RDF which will provide a good adsorption capacity for pollutants migrating through the base of the dump. Enrichment of Fe-oxides, Fe-hydroxides and clays in soil will result in a decrease in the trace pollutants in water. Therefore, the level of some heavy metals, e.g. Pb, Cu, Cr, Ni, will be reduced in the water and enriched in the soil.

D4 APPLICATION OF THE CONCEPTUAL MODEL IN GEOCHEMICAL MODELLING

D4.1 BASE CASE AND DISPOSAL OPTIONS

Based on the proposal, three closure management options will be selected for both the slimes dam and the coarse dump.

Potential rehabilitation / closure management options are proposed as follows (*still to be discussed and finalized*):

- Close slimes facility but coarse waste dump is still operational for a period of time
- Selling of the coarse residue deposits, i.e. remove node 3;
- Use of soil cover (engineered ideal cover or simple soil cover, intermediate cover) with required dump reshaping;
- Treatment measures to stabilize and immobilize the hazardous waste material on the dump to acceptable risk levels, e.g. adding lime, clays, etc;
- Separation of slimes from RDF for disposal.

D4.2 GEOCHEMICAL MODELLING

D4.2.1 Approaches for geochemical modelling (Figure D7 and D8).

The approach for geochemical modelling includes the following parts:

- Initial pore water:
- Rain water infiltration
- Mineralogical data
- Transport properties
- Kinetic data; and boundary conditions
- Adsorption and exchange capacity of clay and hydroxides.

The initial pore water quality will be obtained from slimes water. The water quality data will be evaluated against the seepage water quality data.

Rain water quality will be used for the geochemical modelling. 2-5 rain water samples need to be collected for this study.

Mineralogical data will be determined by XRD and microscopic study. Mineral size fractions and mineral separation might be needed to fulfill this task.

Transport properties: Porosity, permeability, moisture content, O₂, CO₂ and methane concentration will be determined or estimated in this study.

Kinetic data: The kinetic rates for mineral reaction will be collected from literature.

D4.2.2 Geochemical models

Kinetic geochemical modelling will be applied to the base case scenario and the 3 rehabilitation / management options. The base case scenario is defined in the paragraph just before Section D3.1 above. The following geochemical models will be applied in this study:

i) Equilibrium model

Initially an equilibrium model will be used for speciation of the aqueous phases and mineral phases, water quality balance and data quality control by use of Phreeqc2.0 (Appelo, 2001; Parkhurst & Appelo, 1999) and Geochemist Workbench (Bethke, 1996).

ii) Kinetic model

A kinetic model will be applied to all the nodes by use of kinetic rates of the minerals in the dump by use of Phreeqc2.0 (Appelo, 2001; Parkhurst and Appelo, 1999) and Geochemist's Workbench (Bethke, 1996).

iii) 1D transport model

A one dimensional transport model, Phreeqc2.0 (Appelo, 2001; Parkhurst and Appelo, 1999), will be used. In each node, cells, i.e. multi-layers will be defined in terms of length, hydraulic conductivity, gas diffusion rates, convection rates and water quality data. The one dimensional transport model will be able to model the kinetic reactions including oxygen diffusion. This is extremely important for long-term water quality prediction.

iv) Mixing model

All nodes will be mixed proportionally based on the volume of water in the nodes by using the Phreeqc2.0 mixing model as well as Geochemist's Workbench. Through this model, the water quality of the final discharged water will be obtained.

Some of the parameters used as input data for geochemical modelling have been summarised from previous work done by the consultant. This information will provide some reference for the geochemical modelling in this project.

v) Adsorption model

Some of the materials in the RDF have strong adsorption and exchange capacity for some trace metals, e.g. Cr, Cu, Mn, Ni, Pb. These materials are:

- Fe-hydroxides
- Fe-oxides
- Clay minerals
- Mica and altered mica enrichment of Fe-hydroxides

There is also a thick layer of soil materials underneath the RDF. The soil layer is enriched in clays and oxides and hydro-oxides. Adsorption and exchange models will be applied in the base case and some of the management options. The adsorption and ion exchange models will be performed using Phreeqc2.0.

The major pollutants, e.g. Cu, Mn, Ni will be modelled using ACT2, of the Geochemist's Workbench software. Activity-activity diagrams will be drawn for these pollutants in order to identify the controlling physico-chemical conditions.

Figure D7: Approaches for geochemical modelling

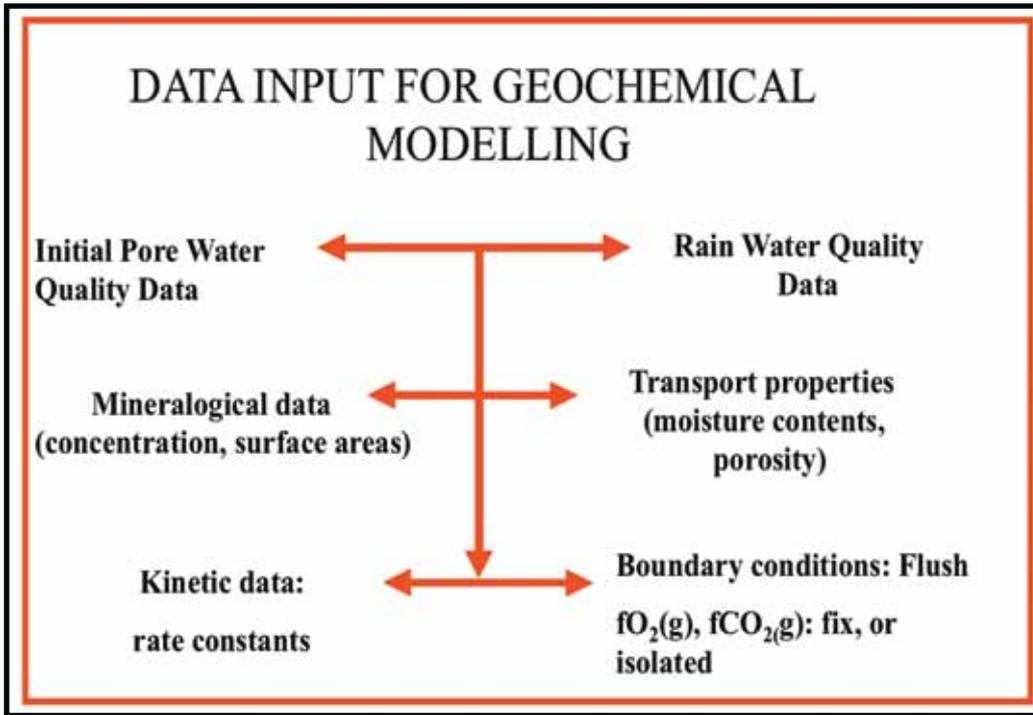
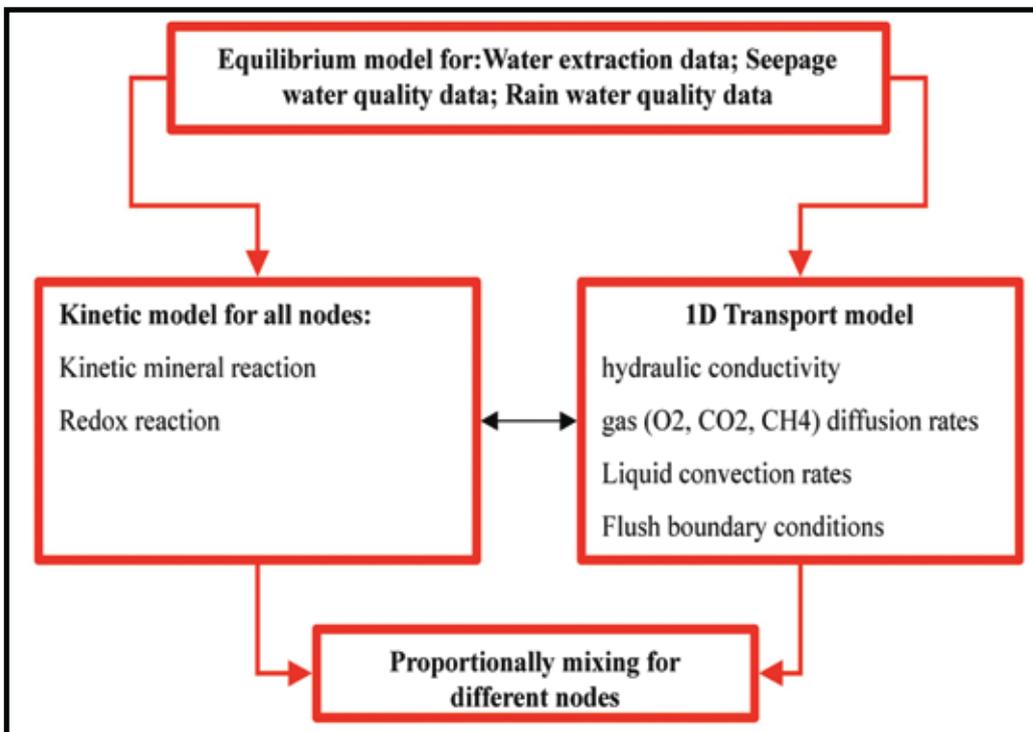


Figure D8: Geochemical models to be applied in this study



**APPENDIX E
EXAMPLE OF CONCEPTUAL MODEL REPORT
FOR TAILINGS DISPOSAL FACILITIES, WASTE
ROCK DUMPS AND RECLAIMED WASTE ROCK
DUMP FOOTPRINTS**

The study requires a detailed conceptual model for each of the following features that will be investigated:

- Tailings disposal facilities;
- Waste rock dumps; and
- Dump footprints.

The constructed conceptual models will be used as guidelines specifically for water balance (hydrology & geohydrology) modelling and geochemical modelling. Simplification is necessary as a complete reconstruction of the field system is neither feasible nor practical from a modelling or assessment perspective.

Water movement and water quality changes through a feature (tailings dam, waste rock dump or dump footprint) are considered in this study. The movement and quality of water once it has left the feature is not within the scope of this study.

E1 TAILINGS DAMS

Von Bredow (1995) suggests that tailings disposal facilities produce a variable and unpredictable quantity of effluent. For this study an attempt is made to put values to these “unpredictable” quantities. Two types of tailings disposal facilities exist in the study areas:

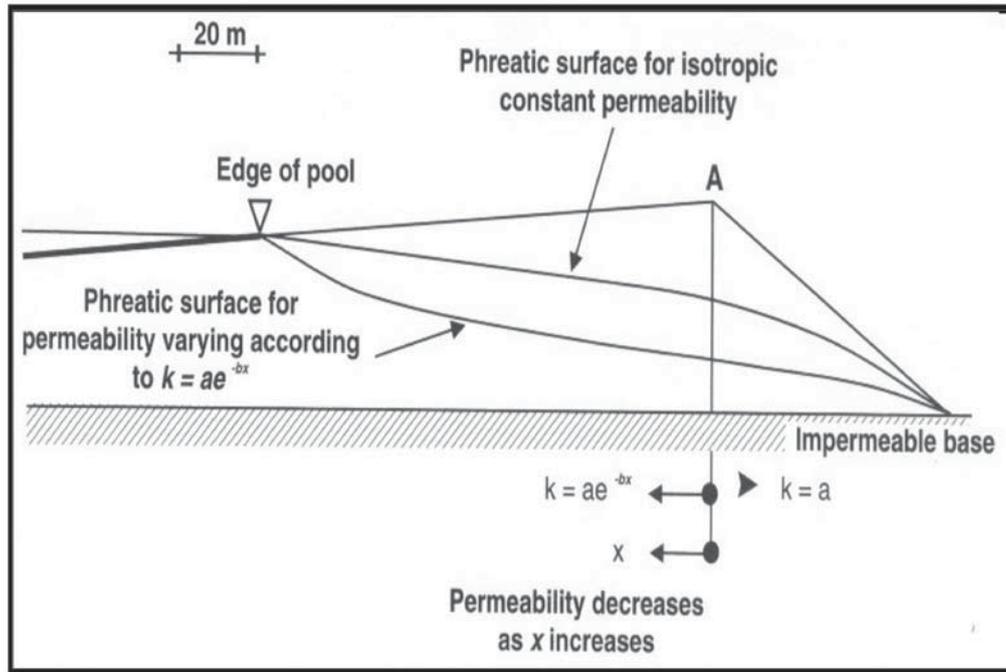
- Operational; and
- Decommissioned.

E1.1 OPERATIONAL TAILINGS DISPOSAL FACILITIES

For this study, long term risks are considered, therefore even the operational tailings disposal facilities will be considered as closed / decommissioned.

Operational tailings differ from decommissioned tailings in that active pool and beach areas exist. Tailings are deposited near the edge of the dam and flow towards the middle of the dam. Due to gravitational sorting a gradient of decreasing permeability exists from the outside to the inside of the tailings dam (Blight, 1987). As a result of this variation in permeability, the phreatic surface of the dam will be depressed (Figure E1). Middleton and Stern (1987) observed the percentage of pool, wet beach and dry beach areas to be 25, 50 and 25 % respectively of the top area of operational dams. International Atomic Energy Agency (1990) suggests that the moisture content of the beach areas above the water table is 10 %, while Middleton and Stern (1987) used a water content of 0.5 ton/ton of residual slimes in their modelling exercise.

Figure E1: Effect of varying permeability on position of phreatic surface in a tailings dam (Chamber of Mines. 1996)



E1.2 DECOMMISSIONED TAILINGS DISPOSAL FACILITIES

Tailings disposal facilities consist of fine-grained particles with the following characteristics:

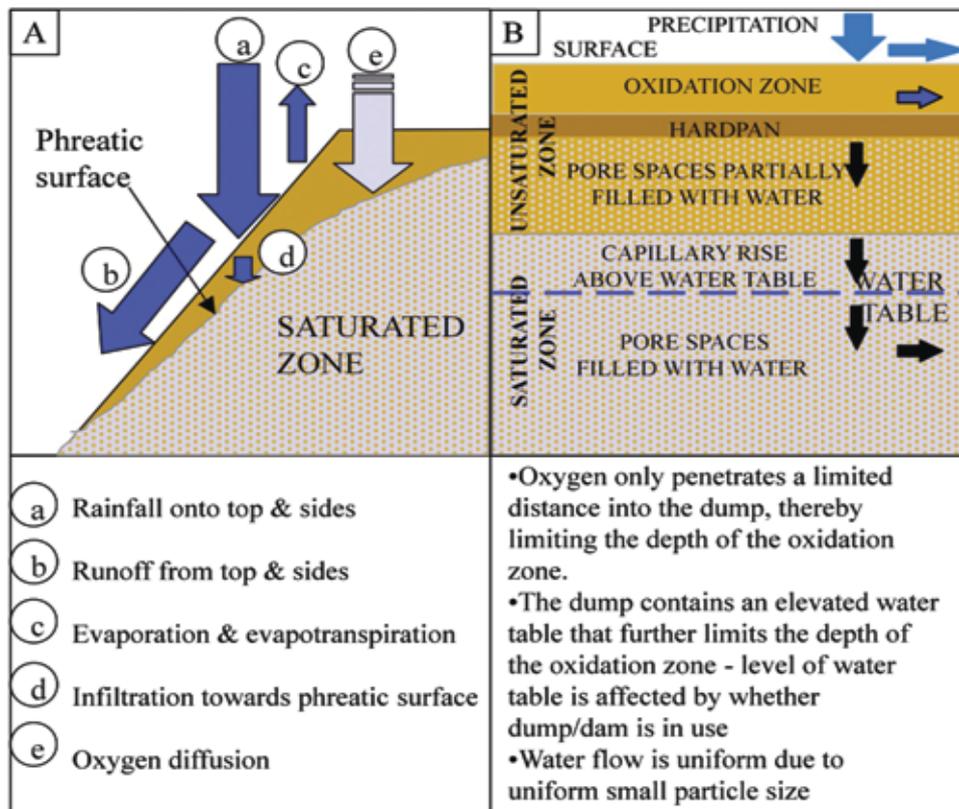
- Particles are relatively uniform in size and exhibit a narrow size distribution.
- The water table within a tailings dam is typically elevated with a portion of the dump being in a saturated state.
- The relative proportion of seepage emanating from the toe of the dump relative to that seeping into the underlying aquifer is dependent on the permeability contrast between the waste material and the underlying geology and the difference between horizontal and vertical permeability of the tailings.

The environment within a tailings dump is less conducive to the production of acid mine drainage than that of a waste rock dump. This is primarily due to the fact that the penetration of oxygen is limited (mainly true for operating facilities receiving large regular inputs of water). However, this is counteracted by the fact that fine waste has a small particle size and, consequently, a large surface area of reactive minerals.

The driving forces and processes that characterise the geochemical behaviour of a fine waste residue deposit and that need to be explicitly considered when assessing the quality of water draining from such a waste deposit are shown schematically in Figure E2 below. This Figure shows that the following macro-scale driving forces need to be considered in a fine waste residue deposit:

- Rainfall onto the top and sides of the tailings dam
- Runoff from the top and sides of the tailings dam
- Evaporation and evapotranspiration from the top and sides of the tailings dam
- Infiltration into the dump along preferential flow paths towards the phreatic surface (water table)
- Diffusion of oxygen into the tailings dam
- Seepage of contaminated water from the tailings dam (either at the toe or through the base).

Figure E2: Processes to consider for tailings disposal facilities



The tailings dam consists of an unsaturated upper zone and a saturated lower zone. The unsaturated zone can typically be subdivided into three separate zones:

- an uppermost oxidation zone where the bulk of the sulphide oxidation will occur;
- a hardpan that effectively reduces the amount of vertical oxygen and water penetration and results in the lateral movement of infiltration water; and
- a lower zone where the pore spaces are partially filled with water and where sulphide oxidation occurs at a reduced rate.

The saturated zone typically extends somewhat above the true water table due to the capillary rise of water into the pore spaces. This saturated zone does not allow for any significant sulphide oxidation although effective neutralisation by base minerals may still occur here.

The depth of the oxidation zone is site specific (e.g. Nicholson et al. November 1997, MEND Project 4.6.5ac, Marsden, 1986, John Easton, 2002, *personal communications*) and is affected by many factors. The major factors are as follows:

- 1) Grain size of the tailings - the coarser the grain size, the deeper the oxidation zone.
- 2) Sulphide consumption - the oxidation zone also can migrate downward with time as sulphide material is consumed.
- 3) Water table change - a trend of increasing oxidation rate with increasing depth to the water table was reported by Elberling and Nicholson (1996).
- 4) Oxygen diffusion rates based on Ficks First Law and Second law (Nicholson et al. (November 1997, MEND Project 4.6.5ac).
- 5) Moisture contents in the pore spaces of the tailings disposal facilities.

Values for the depth of the oxidation zone have been obtained from a literature survey and personnel communications with knowledgeable persons. The results are not consistent and the reported oxidation depths fall within a fairly wide range:

1) 20-40 cm.

According to William Laing (Ore Resource Manager, Crown Mine Recovery, 2002, *personal communications*) the depth of the oxidation zone is in the range of 20-30cm) and for sand dumps can reach to 3-6m which indicates the effect of grain size of the tailing dams.

2) 2 - 3m

According to Marsden (1986), and James and Mrost (1965), the oxidation zone occurs in the top 2 m below the surface, with a maximum depth of 3 m below surface as reported in their study based on old tailings disposal facilities in the Witwatersrand Basin.

3) 5 - 7m

According to Jan Daverson, a geologist (2002, *personal communications*), the depth of the oxidation zone is to 5-7m below surface. He based his observation on colour change of borehole information. John Easton (2002, *personal communications*) also reports that the depth of oxidation zone is between 1.5 and 5m below surface. He also mentions that the depth of the oxidation zone is site specific.

Based on above discussion, 3m will be used for this conceptual model, but this value will be assessed in the light of paste pH and colour data collected in this study.

Flow in a tailings dam is dependent on the ratio of hydraulic conductivity in the tailings to that of the underlying material (Department of Water Affairs and Forestry, 2002). If the hydraulic conductivity of the underlying material is much larger than that of the tailings, water will migrate downward (although very slowly) and seep into the underlying materials. If the hydraulic conductivity of the tailings is of the same order of magnitude as that of the underlying material, the horizontal component of flow will become larger than the vertical flow component and seepage can be expected to occur in close proximity to the toe. According to Krantz (2002, *personal communications*) the hydraulic conductivity of tailings is very low (0.0001 – 0.00001cm/sec or 0.0864 – 0.00864 m/day).

Hydraulic conductivity values for tailings are estimated in a range of 0.001 to 10 m/day and are quoted by some authors as:

- 0.01 m/day (Woyschner *et al*, 1997) – for waste rock and fine tailings mixed.
- 0.01 m/day (Barbour *et al*, 1993) – for thickened tailings of a copper-zinc mine.
- 0.004 m/day – 0.17 m/day by Yanful *et al* (1990) – for fine grained tailings.
- 0.056 m/day (International Atomic Energy Agency, 1990).

- 8.64 m/day (Middleton and Stern, 1987) - vertical permeability of settled slimes.

A hydraulic conductivity of 0.001m/day for tailings will be used as an initial estimate in model runs. This value may be adjusted due to model calibration. Hydraulic conductivity in the underlying soils has been reported as 0.4 m/day for the weathered zone and 0.1 m/day for the dolomitic rocks. The hydraulic conductivity of the weathered zone directly underneath a tailings dam is assumed to be lower than reported, due to compaction by the weight of the tailings. An initial estimate for the hydraulic conductivity of the weathered zone directly underneath a tailings dam is 0.4 m/day.

The range of these values indicates the large variability and uncertainty associated with the hydraulic conductivity of tailings. Van Zyl (1987) suggests that the hydraulic conductivity of tailings can be reduced one order of magnitude or more due to consolidation of deposited slurry. In contrast, Von Bredow (1995) suggests that tailings has the capacity of absorbing larger amounts of water than would be expected when considering the physical characteristics (permeability and porosity) alone. This is due to the spherical character of the quartz particles of tailings, which has a greater porosity than flat shaped clay particles. Von Bredow (1995) further suggests that cracks are filled and refilled with coarser material, which improves vertical drainage. The percentage of rainfall that reaches the saturated zone (infiltration) in the tailings dam has been reported as:

- 17.5 % by International Atomic Energy Agency (1990).
- 24.3 % by Yanful *et al* (1990) – for fine grained tailings.
- 25 % by Shafer *et al* (1994) – for tailings.
- 30 % by Cloggans *et al* (1991) – for tailings.
- 48 % by Woyschner *et al* (1997) – for waste rock and fine tailings mixed.

Again the range of these values indicates a large variability and uncertainty.

Another uncertain parameter is the evaporation from tailings disposal facilities. Van Zyl (1987) suggests that potential evaporation depends on atmospheric conditions, but the actual flux across the upper boundary is limited by the ability of the soil to transmit water from below. In fine active tailings, evaporation effects are large and this causes a drying of the upper surface. This drying of the upper surface, in turn, causes an upward flux from the water table. Evaporation values used in the literature are in a range of 15 to 60%. Examples are given as follows:

- 63.5 % of rainfall (for uranium tailings with no cover, International Atomic Energy Agency, 1990)
- Approximately 3 times more than rainfall for an operational tailings dam (Von Bredow, 1995)

- Approximately 2.5 times more than rainfall for an operational tailings dam (Middleton and Stern, 1987)
- 55.5 % of rainfall (actual evaporation measured for grasslands, Everson, 2001)

Rössner (1999) suggested that the phreatic surface subsides by approximately 0.3 m/year. This means that 10 years after decommissioning, the phreatic surface will be 3 m below the surface of the tailings dam.

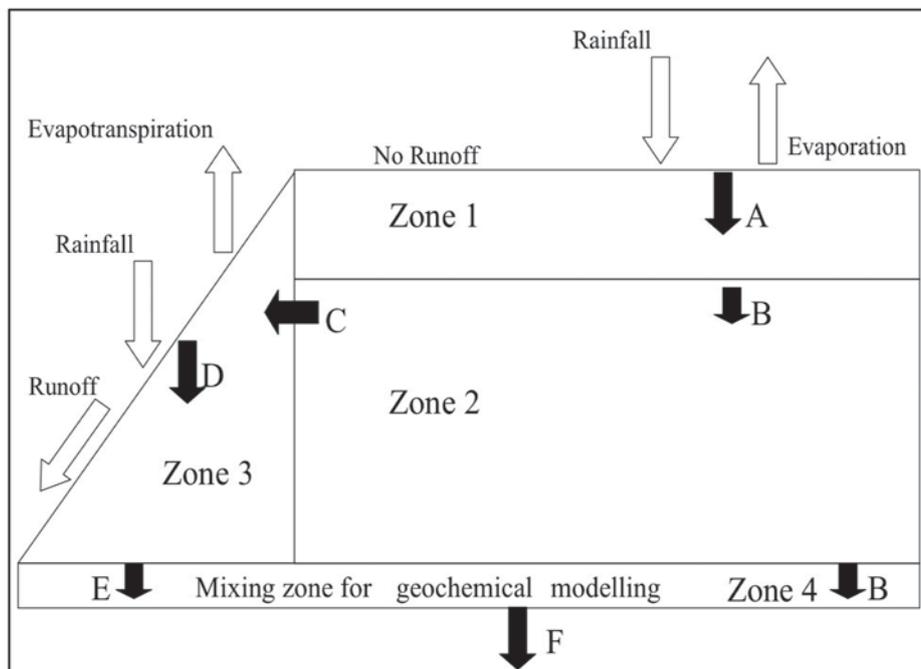
Porosity is the void space between soil particles. The porosity of well sorted soils (such as tailings) is high (Todd, 1980). It can be assumed that the porosity of tailings is 40% which is within the range for other fine grained materials like silt and clay (Chow *et al*, 1988).

E1.3 SIMPLIFIED CONCEPTUAL MODEL FOR TAILINGS DISPOSAL FACILITIES

For modelling purposes, a simplified conceptual model will be used. Four zones will be considered for water movement (Figure E3):

- Zone 1 – the unsaturated zone.
- Zone 2 – the saturated zone.
- Zone 3 – the slope zone (this zone will be assumed unsaturated).
- Zone 4 – the mixing zone for geochemical modelling.

Figure E3: Conceptual model for tailings disposal facilities



The simplified water balance for the conceptual tailings model can be read from Figure E3 as follows:

$$A = B + C$$

$$E = C + D$$

$$F = B + E$$

E1.4 ASSUMPTIONS MADE FOR HYDROLOGY OF ZONE 1

- Runoff is assumed as zero for zone 1.
- The proportion of runoff, evaporation and infiltration will be estimated from the proportions given by

International Atomic Energy Agency (1990). Since the values are given for an entire tailings dam and we are considering the top area only, the proportional values were calculated for each tailings compartment. This resulted in a 22% infiltration and 78% evaporation for zone 1. The infiltration is similar to that used by Yanful *et al* (1990) for fine grained tailings.

- Only vertical flow will be considered for this zone.

Vegetation, irrigation and soil covers on zone 1 can be modelled as rehabilitation options.

E1.5 ASSUMPTIONS MADE FOR GEOCHEMICAL MODELLING OF ZONE 1

- Initial water quality will be that of rain water.
- Air diffusion of O₂ (and CO₂) from the atmosphere to the reduced zone (zero O₂) where Fick's first and second law can be used in the modelling (Marin et al., 1991).
- Major mineral surface areas for pyrite, calcite, quartz, k-feldspar and albite will be used in the geochemical modelling (Parkhurt and Appelo, 1999, Bethke, 1996). The surface area will be determined by analytical methods (Dr Paul de Hault 2002, *personal communications*) and calculation (Anbeek, 1992; White, 1995)
- Initial mineral composition will be taken as the current mineral composition (Parkhurt and Appelo, 1999)
- Parameters for kinetic rates will be taken from the literature (Parkhurt and Appelo, 1999, Bethke, 1996).
- Moisture content will be obtained from analytical results in this study. The preliminary results show moisture content is in a range of 10-20%. The porosity is assumed at (40%).
- A transport model will be used for diffusion of O₂ and CO₂ (Parkhurt and Appelo, 1999, Bethke, 1996). Henry's law will be used to calculate the gas contents in liquid and vapour phases.

E1.6 ASSUMPTIONS MADE FOR THE HYDROLOGY OF ZONE 2

- Water enters this saturated zone from zone 1.
- Initial estimates for the hydraulic conductivity of tailings is 0.01 m/day horizontally and 0.001 m/day vertically.
- Initial estimates for the hydraulic conductivity of the weathered zone directly underneath the tailings dam is 0.1 m/day horizontally and 0.01 m/day vertically.
- Hydraulic conductivity of the weathered zone surrounding the tailings dam is 0.4 m/day (AngloGold, 2000, *personal communications*) horizontally and 0.04 m/day vertically.
- Hydraulic conductivity of the dolomitic rocks is 0.1 m/day horizontally and 0.01 m/day vertically.
- No evaporation takes place from this zone. This assumption is based on the idea that when the water table is more than 3 m below the surface, no upward flux occurs (van Zyl, 1987) and because we are considering decommissioned tailings, the water table

is assumed to be more than 3 m below the surface.

- In this zone the movement of water is distributed between horizontal and vertical components depending on anisotropy.
- It can be assumed that horizontal hydraulic conductivity in this zone is much larger than vertical hydraulic conductivity due to the layered structure and compaction of tailings. Rössner (1999) suggests that while the anisotropy ratio is usually reported to be between 5 and 10 for soils, this value can reach values of more than 200 for tailings. An anisotropy ratio of 200 means that flow is 200 times greater in the horizontal direction than in the vertical direction. The anisotropy ratio will be determined by model calibration, starting with an initial estimate of 100.
- The proportion of flow that travels horizontally and vertically will be determined by modelling with the MODFLOW saturated groundwater flow model.

E1.7 ASSUMPTIONS MADE FOR GEOCHEMICAL MODELLING OF ZONE 2

- Water quality data from zone 1 will be used as initial water quality for this zone.
- There is no air diffusion of O₂ (and CO₂) in this zone, but aqueous diffusion of O₂ might be applied which suits Darcy's law.
- The following parameters will be used in the geochemical modelling: major mineral surface areas (pyrite, calcite, quartz, k-feldspar, albite) which is the same in this zone as in zone 1;
- Major mineral surface areas for pyrite, pyrrhotite, calcite, quartz, k-feldspar and albite will be used in the geochemical modelling.
- Parameters for kinetic rates (pyrite, pyrrhotite, calcite, quartz, k-feldspar, albite) are available from the literature.
- Water volume will equal the porosity (40%) of the tailings. A transport model will be used for aqueous diffusion of O₂;

E1.8 ASSUMPTIONS MADE FOR THE HYDROLOGY OF ZONE 3

The proportion of runoff, evaporation and infiltration will be estimated from the proportions given by International Atomic Energy Agency (1990). Since the values are given for an entire tailings dam and we are considering the slopes areas only, the proportional values were calculated for each tailings compartment. In this

screening level model, no distinction is made between benches and slopes and a uniform slope is assumed. The values differ from one dam to the next due to the differences in ratio between top area and slope area. The mean values calculated are:

- Evaporation: 44 % of rainfall ranging between 24 % and 63 % of rainfall
- Runoff: 44 % of rainfall ranging between 19 % and 69 % of rainfall
- Infiltration: 12 % of rainfall ranging between 7 % and 17 % of rainfall
- Inflow from zone 2 will be added to the infiltration of zone 3 to determine E in Figure E3.
- Toe seepage is included in seepage to groundwater (E in Figure E3).

E1.9 ASSUMPTIONS MADE FOR GEOCHEMICAL MODELLING OF ZONE 3

- Initial water quality will be the mixing of rain water and the water from zone 1 and the proportion will be determined by modelling and estimation.
- Air diffusion of O₂ (and CO₂) in this zone will dominate.
- Major mineral surface areas for pyrite, calcite, quartz, k-feldspar and albite will be used in the geochemical modelling.
- Initial mineral composition will be taken as the current mineral composition (from XRD and other analyses).
- Parameters for kinetic rates will be taken from the literature.
- Moisture content will be assumed at 50% of the porosity (40%).
- A transport model will be used for aqueous diffusion of O₂;
- A mixing model will be used for reaction in this zone.

E1.10 ASSUMPTIONS MADE FOR GEOCHEMICAL MODELLING OF THE MIXING ZONE

- There is no seepage from the tailings dam into surface water sources.
- The different water qualities and volumes from zones 2 and 3 will be mixed in proportion.
- The final water quality will derive from this reaction.

E2. WASTE ROCK DUMPS

Coarse waste residue deposits typically provide an environment that is very conducive to the production of acid rock drainage. This is primarily due to the fact that coarse waste deposits are very permeable and allow the ingress of oxygen to considerable depths. However, this is counteracted by the fact that, per definition, coarse waste has a large particle size, thereby ensuring that a large amount of the minerals that contribute to the change in water quality, are locked up within the larger particles.

In these coarse waste deposits, water may flow in channels and macropores somewhat independently of the hydraulic conditions in the smaller pores. The saturated hydraulic conductivities of the macropore region can be up to several orders of magnitude higher than the hydraulic conductivity of the porous matrix (López *et al*, 1997).

Newman *et al* (1997) suggest that samples where the portion of fine material in a waste rock dump was larger than 40 %, were capable of retaining water under negative pore-water pressures. This capacity to retain water was very small for samples with less than 40 % fine material.

The driving forces and processes that characterise the geochemical behaviour of a coarse waste residue deposit and that need to be explicitly considered when assessing the quality of water draining from such a waste deposit are shown schematically in Figure E4 below.

Figure E4: Processes to consider for waste rock dumps

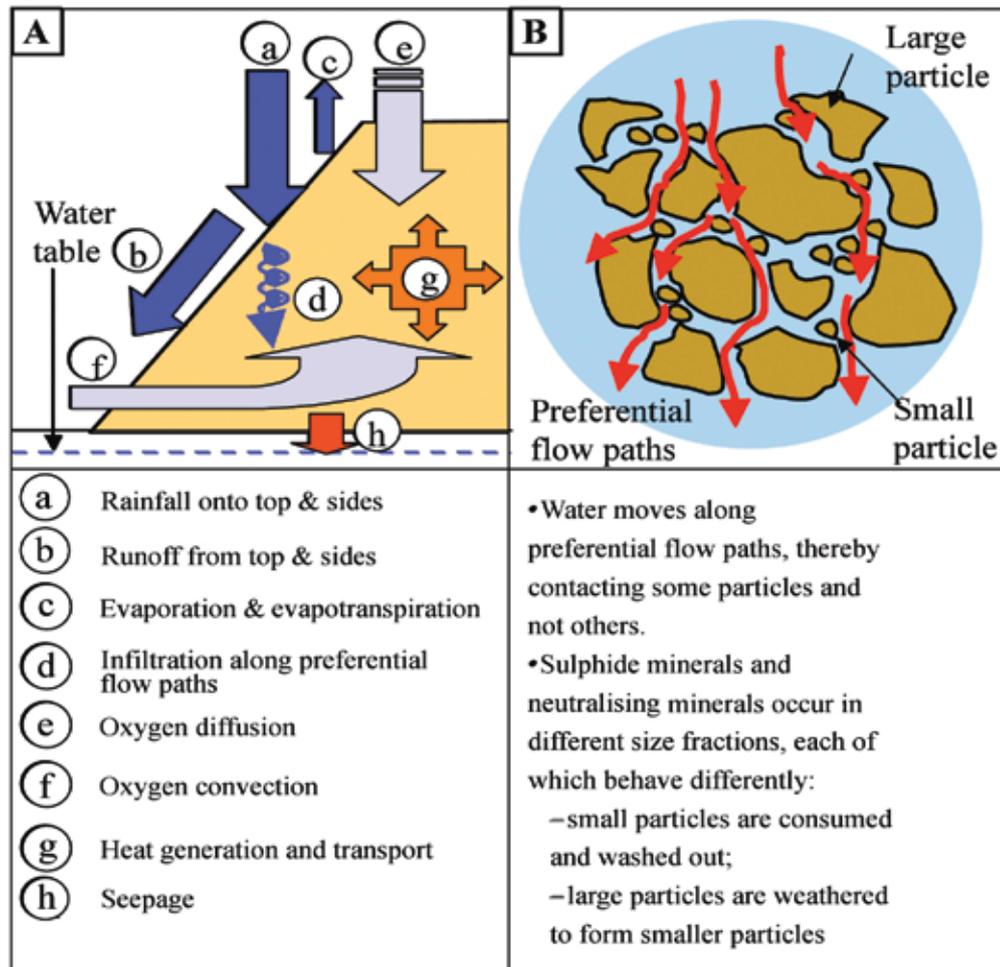


Figure E4 shows that the following macro-scale driving forces need to be considered in a coarse waste residue deposit:

- Rainfall onto the top and sides of the dump
- Runoff from the top and sides of the dump
- Evaporation and evapotranspiration from the top and sides of the dump
- Infiltration into the dump along preferential flow paths
- Diffusion of oxygen into the dump

- Convection of oxygen into the dump
- The generation of heat within the dump (from sulphide mineral oxidation) and the transport of this heat into and out of the dump
- Seepage of contaminated water from the dump (either at the toe or through the base)

The phreatic surface within the coarse waste residue deposit is dependent on the volume of water ingress on the dump surface and the permeability of the underlying weathered material/aquifer relative to that of the waste

material. It is anticipated that the underlying lithologies generally have a lower permeability than that of coarse waste residue deposits.

Since the permeability of the coarse waste material is relatively high, one would expect the phreatic surface within the dump to be relatively flat and located within the lower portion of the dump. This depressed water table has two very important effects that need to be considered (Department of Water Affairs and Forestry, 2002):

- Water flows vertically down through the dump along preferential flow paths - this means that some particles are fairly isolated from the water flow path.
- Oxygen penetrates deep into the dump and the spaces between particles are typically filled with air (and oxygen), while particles are surrounded by a thin layer of water.

Leachate from a coarse waste residue deposit is most likely to seep from the face of the dump in close proximity to the toe rather than percolating into the underlying aquifer. This is attributed to the relatively low permeability of the underlying material which cannot accommodate the seepage volume from the coarse waste residue for any given rainfall event.

Uncontrolled leachate that emanates from the coarse waste residue as surface runoff will flow downgradient until it percolates into the weathered soil profile. The distance that such a surface runoff covers before seeping into the underlying aquifers is dependant on the seepage volume, the permeability of the underlying material and the topographic slope in the immediate vicinity of the coarse waste residue. The vertical percolation of the leachate into the underlying aquifers may be hindered by the upward convergence of groundwater in the immediate vicinity of water courses. However, relatively clean groundwater could come into contact with mine waste residue as a result of the convergence process. This contaminated groundwater will usually enter the water courses as baseflow (Department of Water Affairs and Forestry, 2002).

E2.1 SIMPLIFIED CONCEPTUAL MODEL FOR WASTE ROCK DUMPS

The simplified conceptual model for water movement through the waste rock dumps considers three zones (Figure E5). The reason for three model zones as opposed to four for tailings, is that the hydraulic conductivity of a waste rock dump is much higher than for tailings and the saturated zone will be much lower and relatively flat. For

the screening-level assessment, the whole of the waste rock dump is considered to be unsaturated.

The simplified water balance for the conceptual tailings model can be read from Figure E5 as follows:

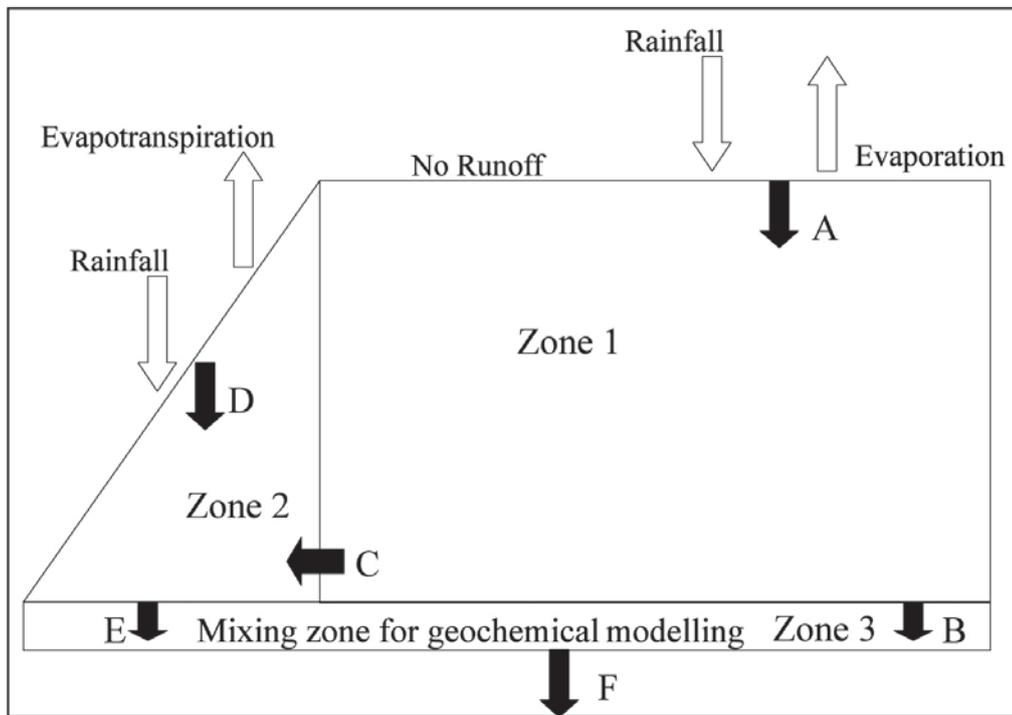
$$A = B + C$$

$$E = C + D$$

$$F = B + E$$

The percentage of fine material (<2 mm) in the waste rock dumps as determined from sampling is 39 % for dump number 4 and 38 % for dump number 3. This is marginally lower than 40 % of fine material and therefore the water retaining capability of these dumps is considered to be small (Newman *et al*, 1997). This implies that infiltration will be relatively high and runoff very low. The slopes (37%) of the waste rock dumps may contribute to runoff under high intensity rainfall events. Daily rainfall data from Weather Bureau station 04362972 at the mine suggests that it rained more than 10 mm/day for 5% of the time between January 1990 and June 2002. Rainfall events of more than 23 mm/day occurred 2 % of the time, which is approximately 7 days a year. This indicates that high rainfall events are rare and they will be ignored in this simplified model. An average rainfall of 2 mm/day will be used.

Figure E5: Conceptual model for waste rock dumps



Hydraulic conductivity of a waste rock dump can be up to several orders of magnitude higher than the hydraulic conductivity for the porous matrix (López *et al*, 1997). Todd (1980) suggests hydraulic conductivity values of 45 m/day and 150 m/day for coarse sand and coarse gravel respectively. The hydraulic conductivity of the waste rock dump will be assumed to be within this range.

In poorly sorted soils such as the material of a waste rock dump, porosity is low (Todd, 1980). The reason for this is that finer particles move into the void spaces between larger particles. Porosity of waste rock dumps will be assumed to be 35% for modelling purposes.

The slope area of waste rock dumps 3 and 4 contribute to 73 % and 81 % respectively of the total area of the dumps.

A conceptual model has been developed to calculate evaporation from a waste rock dump. The following assumptions were used:

- Water available for evaporation occurs in the top 200 mm of the waste rock dump.
- Any water that infiltrates deeper than 200 mm will ultimately report as seepage.

- Assume that surface moisture that can be retained in the 200 mm layer is equivalent to 6.5 % of the mass of waste rock in this layer.

From the equation below, the mass of water available for evaporation was calculated. This mass was converted to a volume of water (1000 kg of water = 1 m³). By dividing this volume of water by the footprint area of the waste rock dump, the mm of rain required to produce the volume of water stored in this 200 mm layer can be calculated.

$$\text{Mass of water} = A * T * (1 - P) * SG * SM$$

Where:

- A = Entire area of waste rock dump (in m²).
- T = Thickness of layer from where evaporation can take place (0.2 m)
- P = Porosity (0.35), 1-P = 0.65
- SG = Specific Gravity (2 700 kg/m³)
- SM = Surface moisture (6.5%)

Using this method, the amount of rainfall needed to fill the storage space that can contribute to evaporation is 13.5 mm for # 3 rock dump and 15.2 mm for # 4 rock dump. Daily rainfall data for the Weather Bureau station from January 1995 to December 2001 were used to determine rainfall events of more than 13.5 mm/day. Single events

of more than 13.5 mm of rain per day occurred 110 times in the period 1995 to 2001 or approximately 16 times a year. Rainfall events of more than 13.5 mm over a period of five days with less than 13.5 mm/day occurred only 7 times in the 7 year period, and therefore it was decided to ignore cumulative rain over a period of time.

The following procedure was used to calculate infiltration from daily data:

- If rainfall for a day is less than 13.5 mm, there will be no infiltration
- If rainfall is more than 13.5 mm, the amount of rain that is more than 13.5 mm will infiltrate

This resulted in a mean infiltration rate of 0.67 mm/day which is 38 % of the mean daily rainfall. Although this may appear to be low infiltration for a waste rock dump, this value is almost double the infiltration assumed for tailings disposal facilities and approximately ten times more than the recharge used for Karoo aquifers (Kirchner and van Tonder, 1991). Evaporation from waste rock dumps will then be 62%.

Assumptions made for the hydrology of zone 1

- There is no vegetation on top of the dumps.
- There is no runoff from this zone.
- Based on the rapid response of the water table to rainfall events (López et al, 1997) and the fact that flow is mainly unsaturated, it is assumed that flow in a waste rock dump is mainly vertical.
- When the infiltrated water reaches the underlying soils, flow will divert horizontally due to lower hydraulic conductivity of the underlying soils. This will cause seepage near the toe of the waste rock dump.
- A small horizontal flow component is caused by preferential flow paths, which are not strictly vertical.
- Evaporation is assumed to be 62 % of rainfall.
- Infiltration is assumed to be 38 % of rainfall.

Assumptions made for geochemical modelling of zone 1

- Initial water quality will be that of rain water.
- Air diffusion of O₂ (and CO₂) vertically from the atmosphere to ground surface (zero O₂) where Fick's first and second law can be used in the modelling (Marin et al, 1991);
- Major mineral surface areas for pyrite, calcite, quartz, k-feldspar and albite will be used in the geochemical modelling.

- Initial mineral composition will be taken as the current mineral composition.
- Parameters for kinetic rates will be taken from the literature.
- Moisture content will be obtained from analytical results. The preliminary results indicate that the moisture contents are in a range of 10-15%. The porosity is assumed at 35%.
- Transport modelling will be used for diffusion of O₂ and CO₂.

Geochemical modelling of this zone (and also zone 2) will incorporate the following steps:

- Initial data input of water quality data (run water), and minerals;
- Kinetic module, factors of time reaction rates of mineral rates and time will be built in;
- Transport module (O₂, CO₂)
- pH calculation;
- Major elements calculations;
- Possibly major pollutants (trace metals);
- Acidity module

Assumptions made for the hydrology of zone 2

- No vegetation occurs on the slopes of a waste rock dump.
- No runoff is assumed for average rainfall conditions.
- Evaporation is assumed to be 62 % of rainfall.
- Infiltration is assumed to be 38 % of rainfall
- Seepage E in Figure A5 includes seepage to the side of the dump.

Assumptions made for geochemical modelling of zone 2

- Initial water quality will be that of rain water and the water from Zone 1.
- Fick's first and second law regarding air diffusion of O₂ (and CO₂) from the atmosphere to ground surface (zero O₂) will be used in the modelling (Marin et al., 1991).
- Aqueous diffusion of dissolved gases for waste rock (transport model) accounts for the transport of O₂ through water to acid-generating minerals on rock particles.
- Major mineral surface areas for pyrite, calcite, quartz, k-feldspar and albite will be estimated and used in the geochemical modelling.

- Initial mineral composition will be taken as the current mineral composition (from ABA test, XRD and other analyses).
- Parameters for kinetic rates will be taken from the literature.
- Moisture content will be obtained from the analytical results and the porosity is assumed at 35%.

Geochemical modelling will follow a similar process as indicated for Zone 1.

Assumptions made for geochemical modelling of the mixing zone

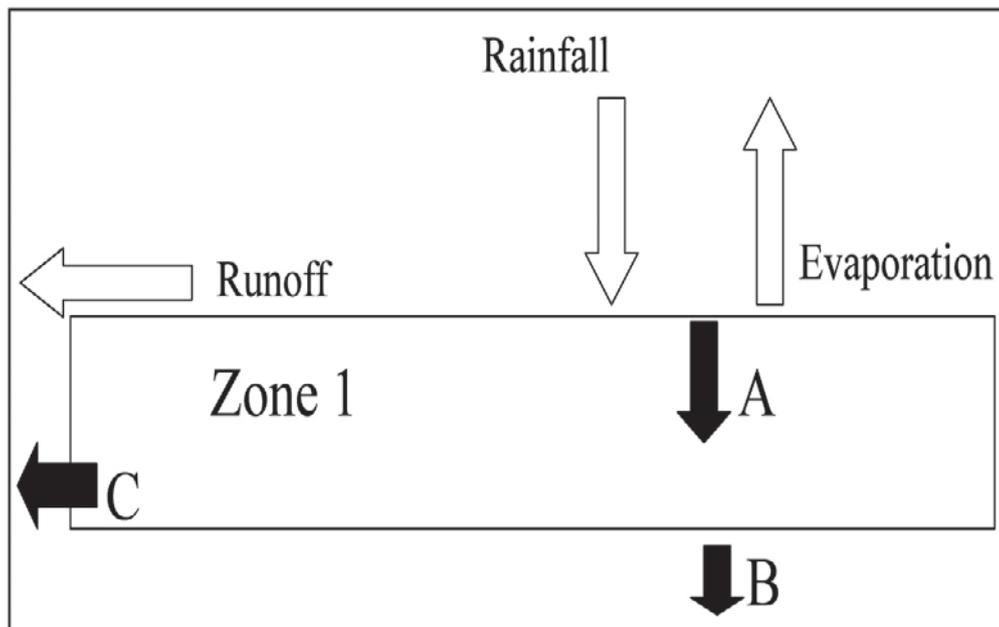
- Mixing of ground waters from zone 1 and zone 2.

A mixing geochemical model will be applied to this zone.

E3. DUMP FOOTPRINTS

Figure E6 shows the conceptual model for dump footprints.

Figure E6: Conceptual model for dump footprints



E3.1 SIMPLIFIED CONCEPTUAL MODEL FOR DUMP FOOTPRINTS

Footprints will be modelled in a similar fashion to zone one of the tailings disposal facilities. From the report by Lake and Hattingh (2001) it is clear that every site is different and flow is dependent on the geology of underlying material. They describe different sites with:

- Predominantly horizontal flow.
- Predominantly vertical (downward) flow.
- Predominantly vertical flow until clay layers are encountered when horizontal flow begins to exceed the vertical component.

Assumptions made for the hydrology of dump footprints:

- Footprints are contaminated up to a depth of 0.5 m (Lake and Hattingh, 2001)
- Flow is predominantly horizontally in the weathered zone
- Runoff from the footprint is related to surface topography.
- Evaporation is assumed to be that of grasslands measured by Everson (2001) as 55 % of rainfall.
- Infiltration is assumed to be 6 % of rainfall (Krantz, 2002, *personal communications*).

- Infiltration is dependent on the geology of the underlying material.
- A hydraulic conductivity of 0.4 m/day was measured in the upper weathered permeable zone and 0.1 m/day for the dolomitic rocks.
- Hydraulic conductivity for the compacted area of residual tailings is assumed to be 0.01 m/day horizontally and 0.001 m/day vertically.
- The contaminant migration rate is 20 m/year (=0.05 m/day) (Mining Company, 2002, *personal communications*).

Assumptions made for geochemical modelling of dump footprints

- Initial water quality will be that of groundwater.
- Flush water is assumed to be rain water.
- XRF will determine the contamination of the footprint due to chemical leaching from the waste rock.
- Pollutants are stored in clay minerals and pore water through adsorption.
- Transport and retardation models will be used.
- Kinetic modelling might be employed to assess water quality if any sulphides and carbonates are left in the waste rock footprint.

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