Risk based decision tool for managing and protecting groundwater resources

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Decision Tool for establishing a Strategy for protecting groundwater resources: Data requirements, Assessment and Pollution Risk

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Disclaimer

The authors accept no responsibility and liability for use of the decision tool. Including without limitation fitness for a particular purpose, functionality and data integrity.

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The steering committee responsible for this project, consisted of the following persons:

Mr K Pietersen	Water Research Commission (Chairman)
Mr HM du Plessis	Water Research Commission
Mr EC Murray	CSIR
Dr Y Xu	Univeristy of the Western Cape
Mr L Botha	Jasper Müller and Associates
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EXECUTIVE SUMMARY

BACKGROUND AND MOTIVATION

Water of acceptable quality is both necessary for the improvement of the quality of life and essential in the maintenance of all forms of life. The semi-arid and arid regions of South Africa form approximately 66% of the country. Most of these regions do not have surface water resources. Groundwater is therefore becoming an important component of the water supply for many South African rural communities in these regions. It also offers a cost-effective solution for rapidly developing settlements, which lack the necessary infrastructure normally associated with water supply. The availability of water for various uses is directly related to the management of water quantity, quality and/or the elimination of diseases.

This report presents the findings of research conducted by the Institute for Groundwater Studies on the development of a risk-based decision tool (DT) to manage and protect groundwater resources. A risk can be defined broadly as the probability that an adverse event will occur in specified circumstances. Effective decision-making involves the management of risks: the identification, evaluation, selection and implementation of actions to reduce risk. Risk assessment is a technique that provides such information to the manager, thereby facilitating the complex and integrated decisions required.

STATEMENT OF OBJECTIVES

The objectives of the project discussed in this document are summarised as:

- To develop a preliminary guidance framework for early stage risk assessment prior to any new data measurements, considering both the probability and the economical consequences of contamination,
- To provide a basis of cost-effective decision-making regarding groundwater protection and management options,
- To provide a risk assessment framework which optimises use of professional judgement for studies where data are limited,
- To give theoretical values that complement field measurements and that can be used as prior estimates in more detailed studies, and
- To perform tracer experiments at a number of contaminated sources to provide information on parameters like matrix diffusion, dispersivity and the possibility of remediation.

The aim of this research is to develop a DT to aid groundwater resource managers in the task of optimising the utilisation of groundwater. The DT will include:

- *Information concerning aquifer parameters*: Pumping test analysis methods have been developed primarily to investigate and characterise flow within idealised confined radial flow systems. Unfortunately these assumptions are usually invalid with regard to the shallow fractured rock aquifers in South Africa. Notable attempts have been made to expand pumping test methodologies. A worthwhile method to consider when analysing a pumping test was developed by Barker (1988), who generalised the Theis equation by including a term called the non-integer flow dimension, thereby making it applicable to arbitrary fractured confined aquifers.

- *Information concerning contaminant parameters*: Dispersivity is a scale-dependent property of an aquifer that determines the degree to which a dissolved constituent will spread in flowing groundwater. No detailed investigation was conducted concerning this parameter, but as it plays an important role in the movement of contaminated groundwater, it is briefly discussed.
  Although matrix diffusion can influence groundwater contamination, very little research has been conducted on this topic in South Africa. The project therefore includes laboratory matrix diffusion experiments. The results of these experiments are included in the decision tool.
  For the investigation of risk assessment and remediation of groundwater contamination, it is important to estimate transport parameters such as groundwater velocity, effective (or kinematic) porosity and dispersion. For high confidence results, these parameters have to be analysed from field tests, known as tracer tests. As tracer tests under natural conditions, with several observation boreholes, require much time and are costly, different single-well and dual-well tracer tests were explored.

- *A framework for risk assessments:* The project introduces tools based on fuzzy logic to assist in decision-making by systematically considering all possibilities. This tool takes into account the sustainability of a groundwater resource, the potential contamination of groundwater, human health risks and the impacts of changes in groundwater (quantity and/or quality) on aquatic ecosystems. The risk-based framework optimises the use of professional judgement and includes a database of interpretative values and parameters that can be used prior the any field investigations.

- *Methods for making cost-effective decisions*: Negative impacts can place heavy burdens on society and economics. Cost-benefit-risk assessments are therefore considered to define, compare and measure benefits and costs with regards to an impact.

- *Possibilities of remediation*: Remediation forms an important component of many groundwater investigations and experiments were therefore conducted, the results of which are included in the decision tool. The results provide the groundwater manager with an indication of the possible success of a remediation project.

METHODS AND SUMMARY OF RESULTS

In this study, a tool that uses fuzzy logic based risk assessments to make decisions influencing groundwater management in South Africa is discussed. As a process of evaluating the potential for adverse impacts, risk assessment provides managers and the public with the means to surpass observations about relationships between events and their effects and, by so doing, to answer questions about what is safe and what is unsafe. However, the priority in performing any risk assessment is clarifying he factual and scientific basis of the risks posed. As such, both qualitative and quantitative evidence regarding the nature of the effects, their severity, and their reversibility or preventability must be examined.

In order to obtain accurate results from the risk assessment process, accurate data must be used. This report sets aside a chapter to discuss both aquifer and contaminant parameters and methods to obtain both sets of parameters.

The DT is divided into three tiers namely a rapid, intermediate and comprehensive assessment. For each of the tiers the following risk assessments can be performed:

- A groundwater risk assessment can be defined as the probability of an adverse effect or effects on the sustainability and/or quality of groundwater associated with measured or predicted hazards.
- A groundwater health risk assessment can be defined as a qualitative or quantitative process to characterise the probability of adverse health effects associated with measured or predicted levels of hazardous agents in groundwater.
- Ecological risks of interest differ qualitatively between different stresses, ecosystem types and locations. A groundwater ecological risk assessment quantifies the impacts of groundwater quantity and quality on ecosystems.

Once the desired risk assessments have been completed, cost-benefit-risk analyses can be used to aid in decision-making regarding the management of a groundwater resource. A cost-benefit-risk analysis is defined as a set of procedures used for defining, comparing and measuring benefits and costs, which originate from either an investment or the operation of an activity.

Since the early 1980's geohydrologists and engineers have developed a number of techniques for protecting groundwater. Protection is divided into two categories: measures to prevent failure and pollution of water resources, and measures to remedy the effects of polluted water resources.

On completion of the different aspects of the DT a report will be generated including the input data and the results of the risk assessments and cost-benefit-risk analysis. Depending on the user, prevention measures can be included. Unfortunately no in-depth study has been completed on remediation options, but the user will be able to browse through the various options.

MEETING THE OBJECTIVES

All the objectives of this project were met. There are many risk assessment methodologies available, however after consulting with Prof George Pinder of Princeton University it was decided to follow a fuzzy logic approach which incorporates the knowledge of professionals in both groundwater and risk assessment fields. Professionals who were consulted are:

- Gerrit van Tonder and Ricky Murray, experts in quantifying groundwater potential
- Brent Usher and Frank Hodgson, groundwater consultants who focus on groundwater quality
- Bettina Genthe a health risk assessor
- Christine Colvin and Dave Le Maitre who have spent many hours investigating impacts of groundwater on the ecology. Both are involved developing methods to incorporate these aspects in the Reserve.

Based on discussions with the above-mentioned experts and extensive literature surveys a risk based DT was developed uniquely for South African conditions.

CONCLUSIONS AND RECOMMENDATIONS FOR FURTHER RESEARCH

Included in the DT are methodologies to characterise fractured rock aquifers. There is ongoing research concerning these aquifers and as new methodologies are developed it is important to include them in the DT.

This DT has been developed over a period of three years and even though it has been tested and calibrated by experts, it is important to note that in order to obtain more accurate results, it must be validated over a period of many years.

In addition the database of the DT has been populated with information; it can be expanded and more detail can be added.

The ecological risk assessment is limited to a few indicators to determine the risks for aquatic ecosystems. This assessment can be developed to include aspects such as flow conditions in rivers and fish species. In addition the impacts of groundwater on terrestrial ecosystems need to be considered and included in the ecological risk assessment. Accommodating these factors complicates ecological risk assessments.

The cost-benefit-risk analysis is crude and this can be developed into more comprehensive computations.

Even though uncertainty has indirectly been included in the DT, further development of the DT should include a comprehensive uncertainty analysis. The uncertainty analysis should include aspects such as the quality of data, the relationships between potential hazards and effects of concern and, the methods used to calculate risks. The uncertainty analysis thereby highlights the limitations of the risk assessment allowing decisions to be made in a more transparent fashion.

The DT developed in this report relies heavily on the expertise of geohydrologists, assumptions and approximations of real world conditions. Together with the heterogeneities present in groundwater systems it is impossible to guarantee the accuracy of the methodologies and the reader must take this into consideration. However as Hurst (1957) stated: It is usually better to do something which is 95% effective immediately, rather than to wait several years to improve the solution by 4%.

The DT can be a useful tool for a groundwater manager to use in order to obtain an understanding of the groundwater situation in a particular area and the impacts thereof. In addition the DT can be used to rank groundwater related problems, thereby making groundwater management and protection an achievable task.

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CHAPTER 1

Introduction

1.1 PREAMBLE

Water of acceptable quality is both necessary for the improvement of the quality of life and essential in the maintenance of all forms of life. The limited number of water resources in South Africa has resulted in increased emphasis being placed on groundwater. Groundwater supply of acceptable quality and quantity is a very important factor in the development of communities. The availability of water for various uses is directly related to the management of water quantity, quality and/or the elimination of diseases. This report introduces a risk-based decision tool (DT) to be used for the management of groundwater.

A risk can be defined broadly as the probability that an adverse event will occur in specified circumstances. Effective decision-making involves the management of risks: the identification, evaluation, selection and implementation of actions to reduce risk. Risk assessment is a technique that provides such information to the manager, thereby facilitating the complex and integrated decisions required. Applications of risk assessments in determining the effects of exposure to contaminants have been institutionalised through legislation in the United States for over 20 years. Other countries such as Japan, Germany, the United Kingdom, the Netherlands and Canada also use some form of risk assessment in decision-making processes (Mazurek, 1996). While there is a growing demand for risk assessments in South Africa, they are yet to become a standard feature (Schwab and Genthe, 1998).

The aim of this research is to develop a DT to aid groundwater resource managers in the task of optimising the utilisation of groundwater. The DT will include:

- *Information concerning aquifer parameters*: Pumping test analysis methods have been developed primarily to investigate and characterise flow within idealised confined radial flow systems. Unfortunately these assumptions are usually invalid with regard to the shallow fractured rock aquifers in South Africa. Notable attempts have been made to expand pumping test methodologies. A worthwhile method to consider when analysing a pumping test was developed by Barker (1988), who generalised the Theis equation by including a term called the non-integer flow dimension, thereby making it applicable to arbitrary fractured confined aquifers.

- *Information concerning contaminant parameters*: Dispersivity is a scale-dependent property of an aquifer that determines the degree to which a dissolved constituent will spread in flowing groundwater. No detailed investigation was conducted concerning this parameter, but as it plays an important role in the movement of contaminated groundwater, it is briefly discussed.

  Although matrix diffusion can influence groundwater contamination, very little research has been conducted on this topic in South Africa. The project therefore includes laboratory matrix diffusion experiments. The results of these experiments are included in the DT.

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- *Methods for making cost-effective decisions*: Negative impacts can place heavy burdens on society and economics. Cost-benefit-risk assessments are therefore considered to define, compare and measure benefits and costs with regards to an impact.

- *Possibilities of remediation*: Remediation forms an important component of many groundwater investigations and experiments were therefore conducted, the results of which are included in the DT. The results provide the groundwater manager with an indication of the possible success of a remediation project.

1.2 SOUTH AFRICAN LEGISLATION

The Constitution of South Africa (Act No 108, 1996) states that everyone has the right to an environment that is not harmful to his or her well-being. It also states that everyone has the right to have the environment protected for the benefit of present and future generations through legislation that prevents pollution and ecological degradation, promotes conservation and secures ecologically sustainable development and use of natural resources while promoting justifiable economic and social development. The Constitution also states that everyone has the right to sufficient water.

During the last few years views on water management and protection in South Africa has changed radically. A prime example of these changes is the New National Water Act (Act No 36, 1998), which focuses on the principles of sustainability and equality. These principles take into account:

- the basic human needs of present and future generations,
- the need to protect water resources,
- the need to share water resources with other countries,
- the need to promote social and economic development through the use of water and
- the need to protect aquatic ecosystems.

Aquatic ecosystems are defined as the abiotic (physical and chemical) and biotic components, habitats and ecological processes contained within rivers and their riparian zones, reservoirs, lakes and wetlands and their fringing vegetation. Terrestrial biota, other than humans who are dependent on aquatic ecosystems, are also included in this definition (DWAF, 1996).

Other legislation that should be considered includes:

- The Water Services Act (Act No 108, 1997): The main objectives of this Act relevant to the research discussed in this document are to provide for:
  - the right of access to basic water supply and the right to basic sanitation necessary to secure sufficient water and an environment not harmful to human health or well-being,
  - the promotion of effective water resource management and conservation
- The Environmental Conservation Act (Act No 73, 1989) provides for the effective protection and controlled utilisation of the environment and for matters incidental thereto.
- The National Environmental Management Act (Act No 107, 1998) regulates co-operative environmental governance by establishing principles for decision-making matters affecting the environment.
- Draft National Health Bill (2001): This Bill promotes the protection, improvement and maintenance of the health of the population.

The final goal of this project is to provide South African groundwater managers with a tool incorporating some of the legislation discussed in this section.

1.3 THE RISK ASSSESSMENT FRAMEWORK *(Summarised from Schwab and Genthe,*

1998)

In this study, a tool that uses risk assessments to make decisions influencing groundwater management in South Africa is discussed. As a process of evaluating the potential for adverse impacts, risk assessment provides managers and the public with the means to surpass observations about relationships between events and their effects and, by so doing, to answer questions about what is safe and what is unsafe. However, the priority in performing any risk assessment is clarifying the factual and scientific basis of the risks posed. As such, both qualitative and quantitative evidence regarding the nature of the effects, their severity, and their reversibility or preventability must be examined. Figure 1-1 is a summary of the risk assessment framework.

Benefits of risk assessments include:

- A clear articulation of the risk. This includes the evaluation of the hazard and the extent and degree of harm that may result. Such an articulation allows risks to be balanced against one another.
- Reveal the uncertainties inherent in the assumption by forcing one to assess the strengths and weaknesses of each assumption in order to estimate the risk by means of the systematic process of a risk assessment. As such, a risk assessment provides a mechanism to allow transparent decisions to be made.
- Inherently flexible. A risk assessment can be targeted to a wide variety of situations and circumstances but can also be tailored to target a specific demographic group, geographic area, temporal period or situation.

There is not any single analytical method for combining information into an estimation of a risk but numerous risk assessment methods that span the spectrum from purely qualitative to highly complex mathematical models. The availability of data, finances and the required outcome will drive the choice of method.

The limitations of scientific information mean that some aspects of the assessment might involve qualitative aspects such as the use of professional judgment. Risk assessments can therefore be seen as a combination of science and judgment.

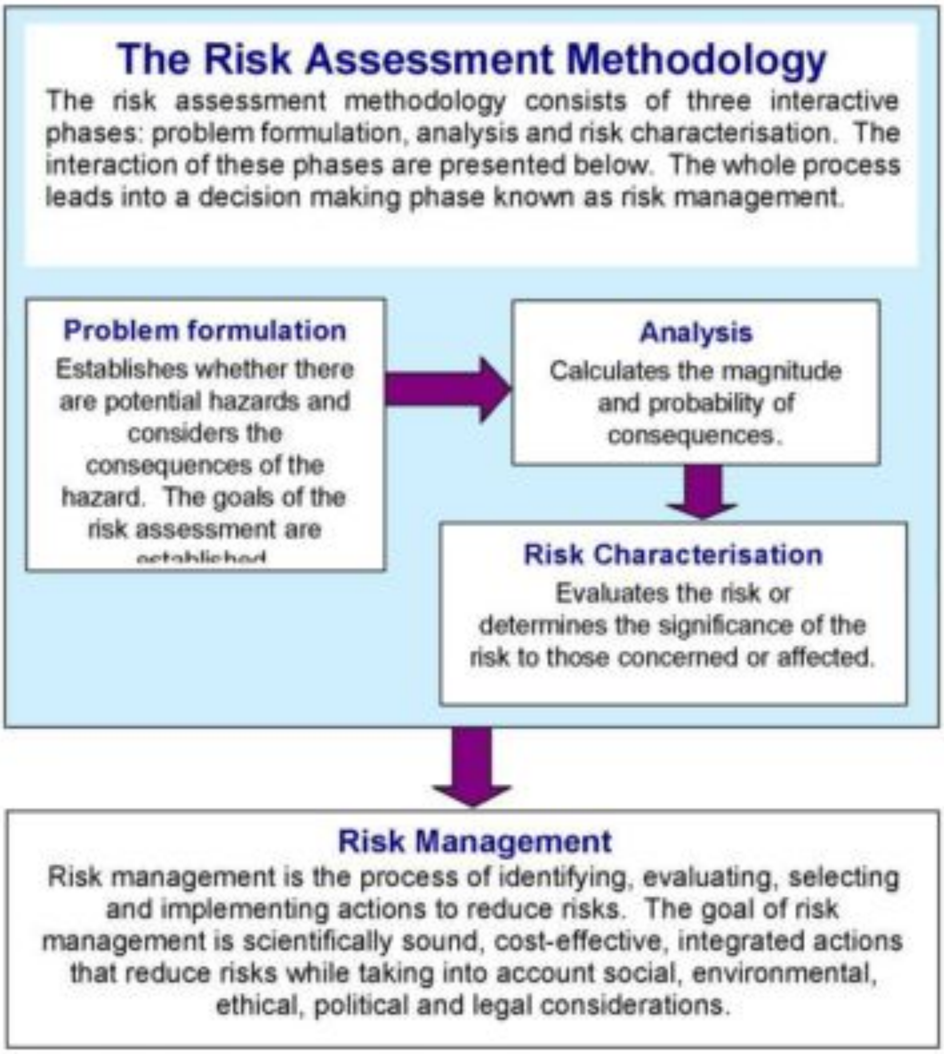


Figure 1-1. Risk assessment framework

1.4 INTERPRETATION OF RISK

Risk assessment is a way of thinking about or analysing a situation, and as such it is a combination of science and judgement. Risk is a combination of two factors: (1) the chance that an adverse event will occur and (2) the consequences of that event. In this report there are four different risk assessments namely:

- Risk of a borehole or groundwater resource failing.
- Risk of a groundwater resource being contaminated.
- Risk of poor groundwater quality affecting human health.
- Risk of an aquatic ecosystem being affected by changes in groundwater quantity and/or quality.

Risk values are stated as a percentage. The higher the percentage the greater the potential of negative impacts. The highest risk obtainable is 99% indicating that under the conditions stipulated in the respective risk assessment there chances of the agent (be it groundwater, human health or an aquatic ecosystem) being impacted are extremely high.

It is the manager's decision as to whether a risk is acceptable or not. This decision must be taken considering both legislation and affected parties. For example a manager might decide a 25% chance of a borehole failing is acceptable, however a 25% chance of a person becoming seriously ill when drinking contaminated groundwater is not acceptable.

The calculated risks are dependent on the confidence in data and method used to calculate the risks, therefore it is important for the manager to understand the fuzzy logic methodology and the associated membership functions.

Risk management is the process of identifying, evaluating, selecting and implementing actions to reduce risks.

1.5 STRUCTURE OF THIS DOCUMENT

The document is divided into 5 main sections:

- The first section (Chapter 2) discusses some new methodologies that can be used to gather information concerning the aquifer and the movement of contaminants. The methods include the analysis of pumping and tracer test data, and the study of dispersivities and matrix diffusion.

- The second section (Chapters 3 – 7) introduces the DT and discusses each of its components. The determination of the risks associated with impacts of natural and anthropogenic activities on groundwater quantity and quality, as well as the potential negative effects on human health, such as infection, toxic effects and the development of cancer, which result from contaminated groundwater, are discussed. As the Water Act (Act No 36, 1998) takes aquatic ecosystems into account, the risks of negative impacts of groundwater (quantity and quality) on aquatic ecosystems are included.

- The third section (Chapter 8) focuses on protecting and remediating groundwater. The protection of water resources is of such importance to the government that a whole chapter of the National Water Act (Act No 36, 1998) is dedicated to this topic. In this chapter, this protection is divided into two categories: measures to prevent the pollution of water resources and measures to remedy the effects of pollution of water resources. These two categories are discussed in this chapter.

- The fourth section (Chapter 9) discusses cost-benefit-risk analyses. Once the desired risk assessments have been completed, cost-benefit-risk analyses can be used to aid in decision-making regarding the management and remediation of a groundwater resource. A cost-benefit-risk analysis is defined as a set of procedures that originate from either an investment or the operation of a service.

- In the last section (Chapter 10), conclusions are drawn and recommendations provided.

CHAPTER 2

Aquifer and Contaminant Parameters

2.1 INTRODUCTION

In order to manage this resource correctly, the geohydrologist has to understand the groundwater system. Abstraction and tracer tests are two of the tools that can aid the geohydrologist in this process. A major objective in cost-effective groundwater protection and management is obtaining optimal value from information obtained from such tests. This chapter will focus on new methods suited for South African fractured aquifer conditions that can be used in the DT.

With increased human settlement and economic development, a number of undesirable substances may find their way into groundwater. It is important for geohydrologists to be able to assess and predict resource pollution. In order to achieve this, they must be able to understand and determine contaminant parameters. Therefore a part of this chapter is dedicated to contaminant parameters and methods to calculate them.

2.2 AQUIFER PARAMETERS

Pumping test analysis methods have been developed primarily to investigate and characterise flow within idealised confined radial flow systems. Unfortunately these assumptions are usually invalid in the shallow fractured rock aquifers in South Africa. The steep increase in drawdown towards the end of a pumping test (Figure 2-1) indicates that the boundary of a fracture has been reached and most of the flow to the borehole is from the matrix. Most analytical methods available today, for example Moench (1984), Cinco-Ley and Samaniego (1981 a&b) and Gringarten et al. (1974), could not be used to analyse this data set, as the methods do not include boundary effects. Notable attempts have been made to expand pumping test methodologies. A worthwhile method to consider when analysing a pumping test was developed by Barker (1988), where he generalised the Theis equation by including a term called the non-integer flow dimension, making it applicable to arbitrary fractured confined aquifers.

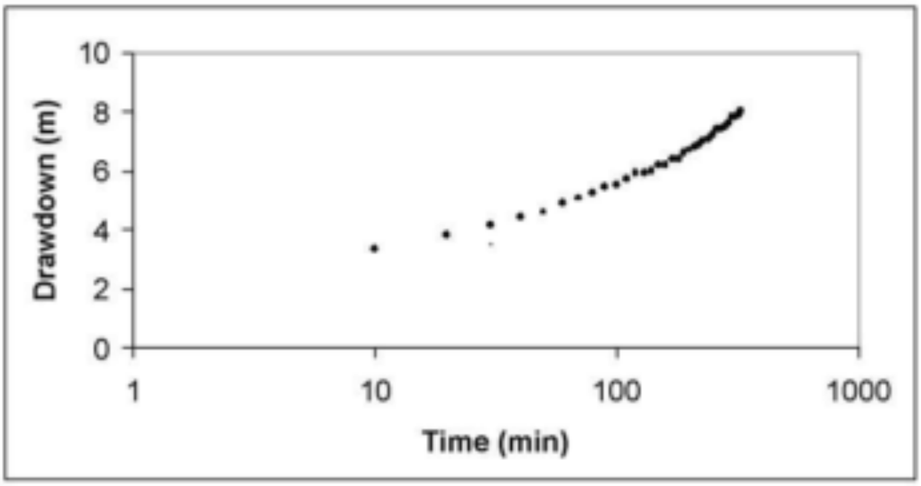


Figure 2-1. Pumping test results from a borehole in the Karoo Sequence. The borehole was pumped at a constant rate of 4L/s.

2.2.1 The Barker Method

One model to describe the flow behaviour in fractured rocks is the generalised radial flow (GRF) model proposed by Barker (1988), which is used to estimate the flow dimension of the fractured aquifer, the hydraulic conductivity and specific storage of the fracture system.

The equivalent system in Barker's GRF-model consists of a homogeneous and isotropic fracture system characterised by a hydraulic conductivity K_f and specific storage S_{sf}, in which the flow to the borehole is radial and n-dimensional. With this model, Barker presents a way of generalising the conventional models used for pumping test analysis for application to arbitrary flow dimensions. For instance, after generalising the Theis equation, it will describe the drawdown in an arbitrary fractured confined aquifer. The generalised Theis equation (Barker, 1988) is written as:

$$s(r,t) = \frac{Qr^{2N}}{4\pi^{1-N}K_f b^{3-n}}\Gamma(-N,u) \tag{2.1}$$

where,

u	=	$r^2 S_{sf} / 4K_f t$
b	=	Extent of flow region (thickness of flow region in case where n = 2)
N	=	N = 1-n/2
n	=	Non-integer flow dimension
K_f	=	Hydraulic conductivity of the fracture system
S_{sf}	=	Specific storage of fracture system
$\Gamma(-N,u)$	=	Incomplete Gamma function

$\Gamma(0,u)$ = $W(u)$ = Theis function

r = The distance along the flow path

If n = 2 (meaning horizontal radial flow to an abstraction borehole) the parameter b is the thickness of the aquifer; for n = 1 (meaning linear flow to an abstraction borehole) the parameter b is the square root of the through-flow area and for non-integer values of n, b has no physical meaning.

It is obvious from Equation 2.1 that there is no unique solution. In order to determine a solution for the above equation it is necessary to fit values for K_f, S_{sf}, b, r and n. The rescaled range method provides a unique method to determine n.

2.2.2 The Hurst Exponent

2.2.2.1 Background (Summarised from Peters, 1996)

The Hurst Exponent (H) was defined by a hydrologist, Hurst, while working on studies of the Nile River. This exponent is also widely used in stock market predictions. H is determined by taking time series data and obtaining the gradient of the plot log(R/S) versus log(i) where

$$R = \mathrm{Max}(X_{t,N}) - \mathrm{Min}(X_{t,N})$$

$$\text{and} \quad X_{t,N} = \sum_{u=1}^{t}(e_u - M_N)$$

with

R = Range of X

$X_{t,N}$ = Cumulative deviation over N periods

e_u = Value of observation at time u

M_N = Average e_u over N periods

t = Time

S = Standard deviation

i = Number of observations

The Hurst exponent can be classified as:

- H = 0.5, which denotes a random data series.
- $0 \leq H < 0.5$, which denotes an anti-persistent time series
- $0.5 < H < 1$, which denotes a persistent time series

H is directly related to the non-integer fractal dimension (D) by D = 1/H.

2.2.2.2 *Hurst Exponent and Pumping Tests*

Pumping test data will normally exhibit persistent behaviour, implying that if the curve has been increasing for a period, it is expected to increase for another period, hence $0.5 < H < 1$. Pumping test data has a long memory component as each observation is correlated to some degree with the observations that follow.

Calculating H is based on a rescaled range analysis. This implies that the data series is divided into N periods, each containing the total number of observation points divided by N. The relationship between the period and the number of observations contained in the period is shown in Figure 2-2. It is important to note that the rescaled range analysis is dependent on the number of observations in a series, and the more observations the more accurate the results. When comparing results obtained for n using Equation 2.1 and then calculating D from the Hurst exponent, it is determined that $D \approx n$ (refer to Figure 2-3).

It is suggested that the following be taken into account to obtain the best results when determining n using Hurst:

- There must be at least 100 data points.
- The data points must be evenly spaced.
- The aquifer must be stressed.
- The observed data must not be noisy. If the data is scattered or noisy a smoothing function is included in the DT to ensure smooth data sets are used in the calculation of n.

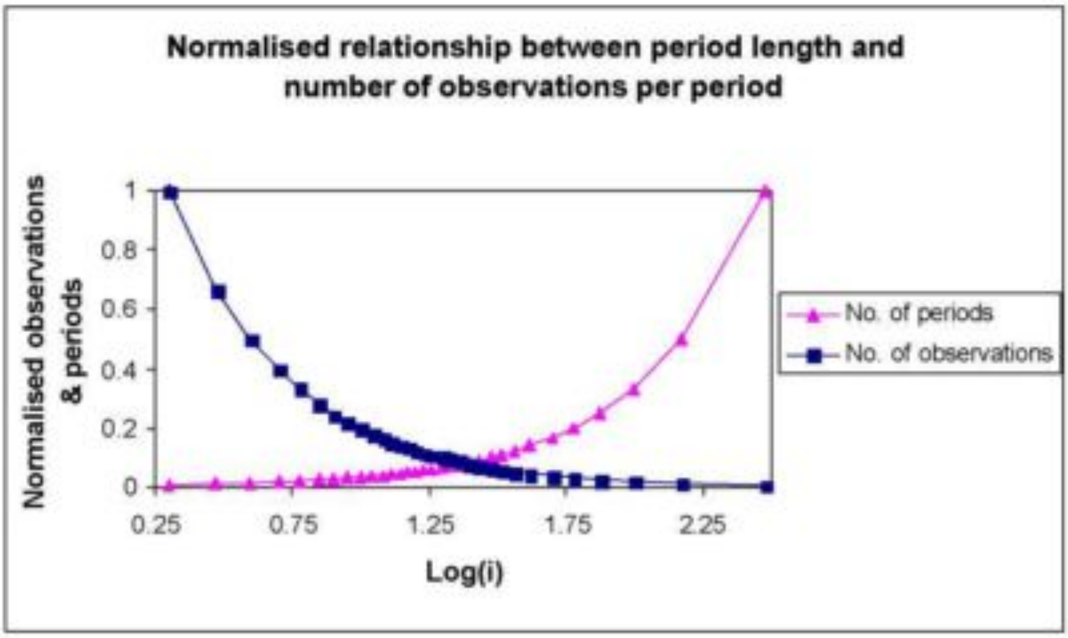


Figure 2-2. Normalised relationship between period length and number of observations per period.

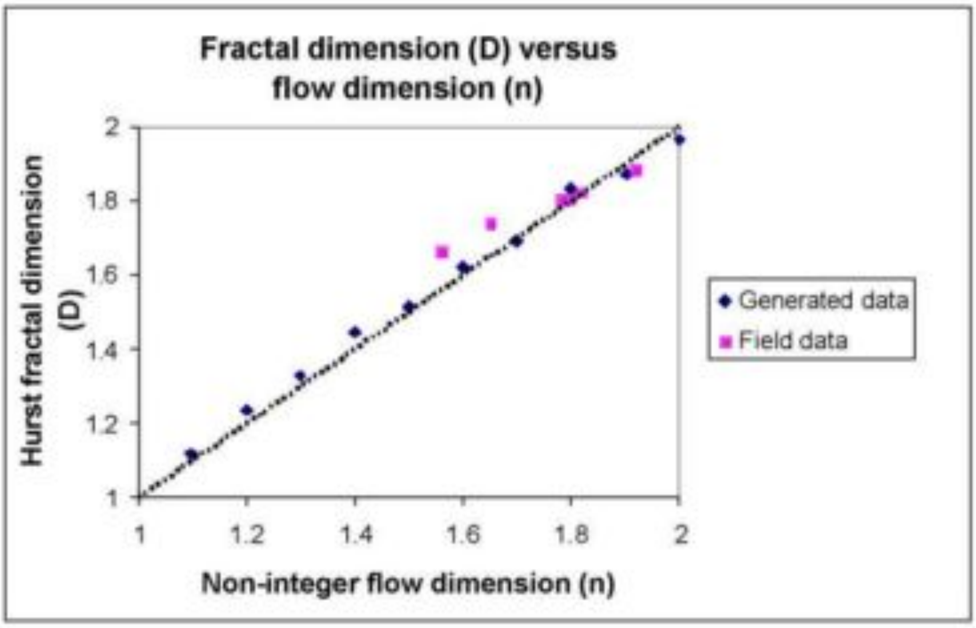


Figure 2-3. Fractal dimension determined using the Hurst exponent versus the non-integer flow dimension calculated with Barker's GRF-model.

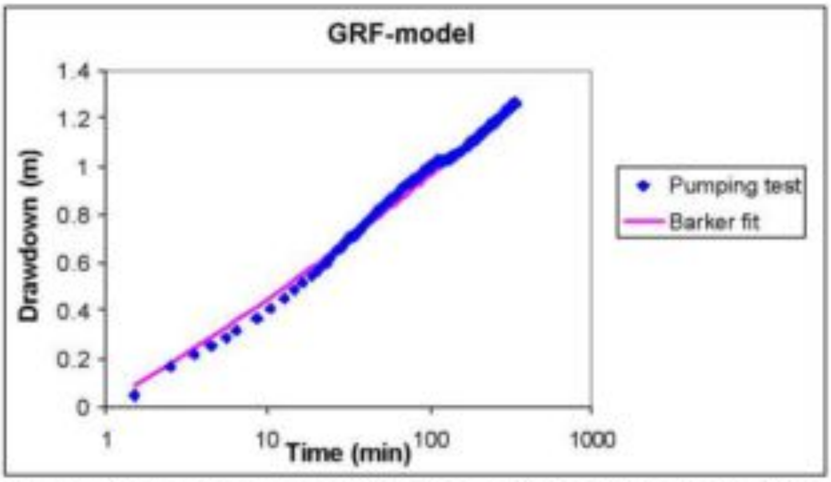
2.2.2.3 *Example*

The Campus test site is located on the grounds of the University of the Free State, South Africa, and covers an area of approximately 180x192 m^2. The thickness of the aquifer on site is approximately 50 m. The aquifer is situated in the Karoo Sequence and the geology consists of sandstone, mudstone and shale deposited under fluvial conditions. Core samples indicate parallel horizontal fractures, the most significant of which is at a depth of 21 m.

Borehole UO5 was pumped for 0.63 L/s for a period of 6 hours. The pumping test data was recorded with a data logger with readings taken every minute. The pumping test data was analysed using both Barker's GRF-model (Figure 2-4) and the Hurst exponent (Figure 2-5).

The non-integer flow dimension determined by Barker's method is 1.85. The fractal dimension D calculated using the Hurst exponent is 1.876.

Figure 2-4. Barker analysis of UO5



It is important for the user to note that the software developed to determine the Hurst exponent is written in such a way that the user will always fit the late time data as shown in Figure 2-5.

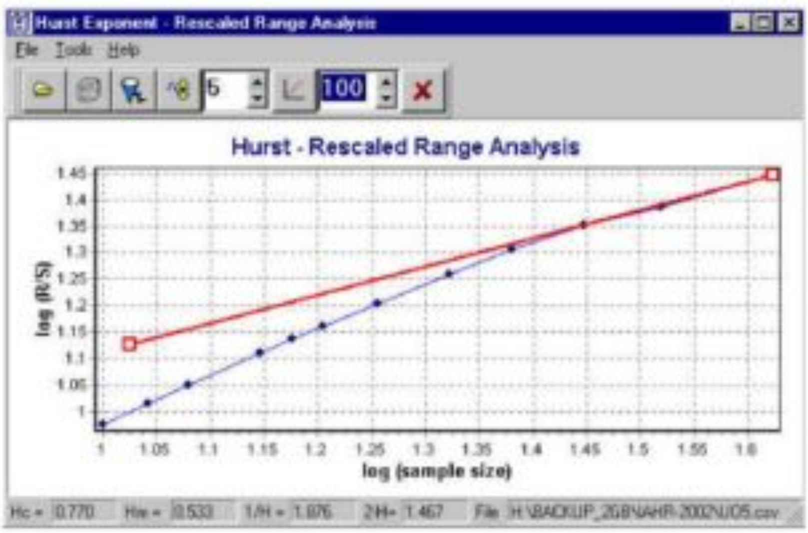


Figure 2-5. Hurst analysis of borehole UO5

2.2.2.4 *Discussion*

The Hurst exponent is calculated from pumping test data and is then inverted to give the non-integer flow dimension. The only drawback of this method is that it is data dependent –

the more observations in a series, the more accurate the results will be. Experimental results indicate that data series with less than approximately 30 observations do not yield accurate results.

For more information concerning the analyses of pumping test data refer to *Manual on Pumping Test Analysis in Fractured-Rock Aquifers* by G van Tonder, I Bardenhagen, K Riemann, J van Bosch, P Dzanga and Y Xu available from the Water Reasearch Commission Private Bag X 03, Gezina, Pretoria, 0031.

2.3 CONTAMINANT PARAMETERS

2.3.1 Dispersivity

Dispersivity is a scale-dependent property of an aquifer that determines the degree to which a dissolved constituent will spread in flowing groundwater. No in-depth investigation was conducted concerning this parameter, although it is important in the movement of contaminated groundwater and as such, is briefly discussed. Figure 2-6 is a graph depicting field-scale dispersivities versus the migration distance, plotted from numerous field measurements. From this graph it can be determined that the relationship between dispersivity and migration distance lies in a zone around the following line:

$$\alpha_L = 0.1 \times L \qquad (2.2)$$

where

α_L = Dispersivity

L = Migration distance of the contaminant

Equation 2.2 is based on a small set of data and will therefore only be used to estimate dispersivities in the intermediate assessments. Low to medium confidence is attached to these results. In Section 2.3.3 a more accurate method for calculating dispersivity is discussed.

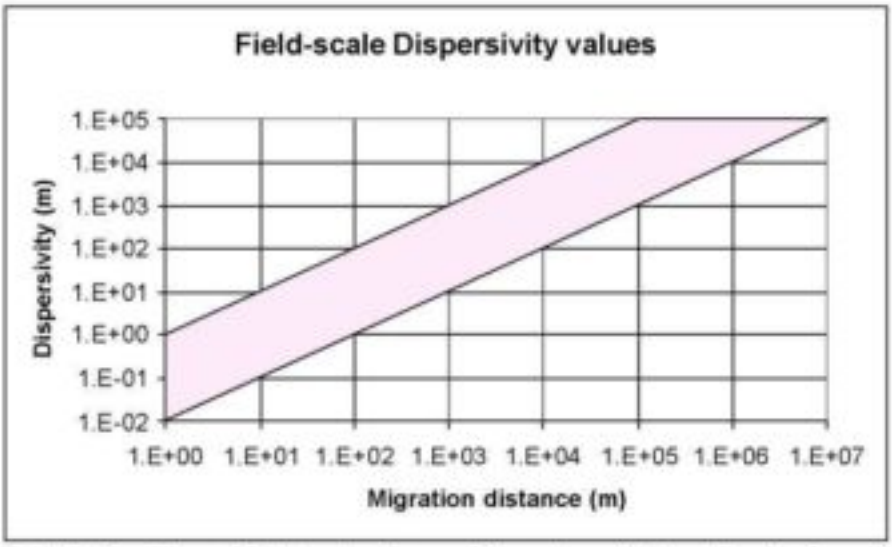


Figure 2-6. Estimation of field-scale dispersivity values *(taken directly from Spitz and Moreno, 1996)*

2.3.2 Matrix Diffusion

2.3.2.1 General

The role of matrix diffusion in groundwater contamination and remediation is in many cases either ignored (Maloszewki and Zuber, 1993) or not fully understood. However more and more geohydrologists are recognising the importance of matrix diffusion (Feenstra *et al.*, 1984; Maloszewki and Zuber, 1993). The aim of the research discussed in this section is to obtain, by means of laboratory experiments, a better understanding of matrix diffusion and the role it plays in the contamination of many of the fractured rock aquifers in South Africa.

Foster (1975) was the first to draw attention to the effects of matrix diffusion on contaminant behavior in fractured rock aquifers. The process of solute diffusion from a fracture to the adjacent matrix is illustrated in Figure 2-7, which schematically shows a constant solute source of constant concentration C_0 transported through the fracture. The effect of matrix diffusion is to provide solute 'storage' with the rate of change in storage within the matrix related to Fick's second law of diffusion. The solute becomes entrapped in the matrix until the concentration gradient reverses. This matrix diffusion results in retardation, causing the bulk of the solute to move at a lower average velocity than the flowing groundwater (Hoag and Price, 1997).

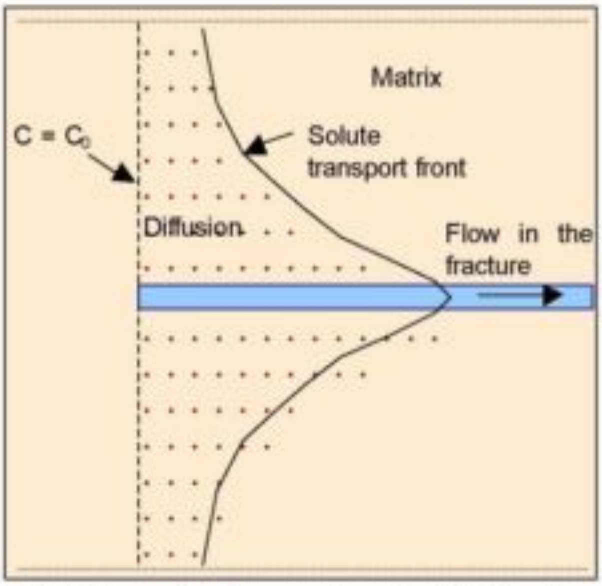


Figure 2-7. Schematic diagram of matrix diffusion in a fracture

2.3.2.2 Methdology

Feenstra *et al.* (1984) discusses a method to determine matrix diffusion in a laboratory. A flat disc of core is dried and then placed in acrylic diffusion cells (Figure 2-8).

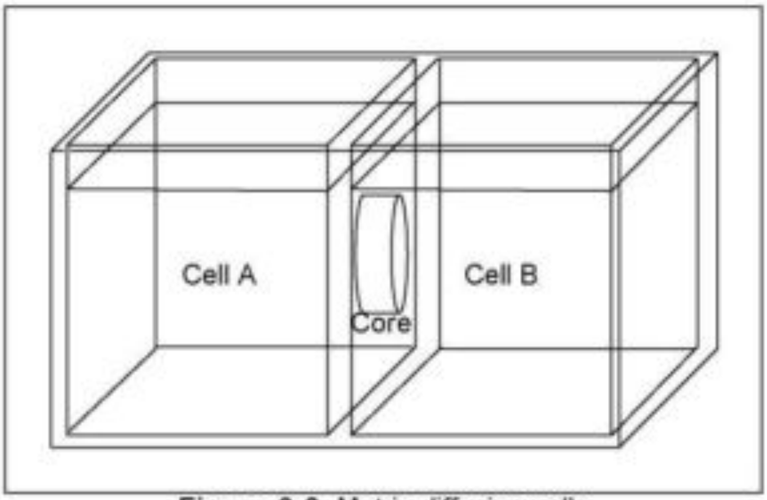


Figure 2-8. Matrix diffusion cells

Deionized water is placed in one of the cells for several days until the water emerges through the sample. Deionized water is then placed in the other cell as well to ensure that the disc is completely saturated. Both cells are then emptied and cell A is filled with a solute, while cell B is filled with deionized water. The diffusion of the solute through the core disc increases the solute concentrations in cell B. The matrix diffusion coefficient can be determined for the core by considering the transfer of mass from one cell to the other through the core disc as (Feenstra *et al.*, 1984):

$$D_{(t_2 \to t_1)} = \frac{(C^B_{t_2} - C^B_{t_1})LV^B}{AC^A(t_2 - t_1)} \tag{2.3}$$

where

$D_{(t2-t1)}$	=	The effective diffusion coefficient measured between time t_1 and t_2
C^A	=	The concentration in compartment A
C^B	=	The concentration in compartment B
V^B	=	The volume of solution in compartment B
A	=	The area of the core disc
L	=	The thickness of the core disc

2.3.2.3 Experiments

A number of matrix diffusion experiments were conducted at the Institute for Groundwater Studies' laboratory in Bloemfontein. These experiments were conducted on sandstones, shales and a quartzite using various concentrations of sodium chloride (NaCl) and sodium sulphate (Na_2SO_4). These experiments will be discussed in the following subsections but a basic experiment will firstly be discussed to familiarise the reader with the methodology.

A basic experiment

The effective matrix diffusion coefficient was calculated for NaCl, which is considered to be a non-reactive solute. The experiment was conducted on a flat disc of fine white sandstone with a porosity of approximately 4.4%. No secondary porosity is evident. The core was 5 mm thick. In order to determine the effective diffusion coefficient, cell A was filled with 10000 mg/L NaCl. Cell B was filled to the same level with deionized water. The diffusion of the NaCl through the sandstone sample resulted in an increase of NaCl concentrations in cell B. The electrical conductivity (EC) values in both cells were measured regularly during the 13-day experiment. The increase in EC values in cell B is shown in Figure 2-9. The diffusion coefficient was calculated using Equation 2.3. The coefficient was calculated for successive time intervals until a constant value was achieved. The value obtained was 3.3 x 10^{-9} m^2/h.

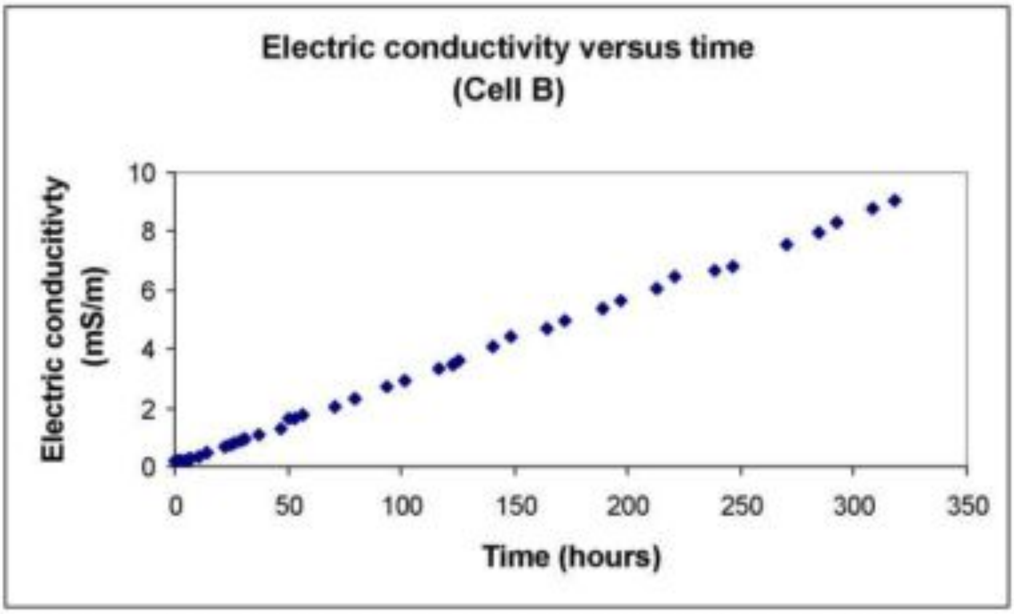


Figure 2-9. Increase in electrical conductivity values with time in Cell B.

Matrix diffusion coefficients and porosity

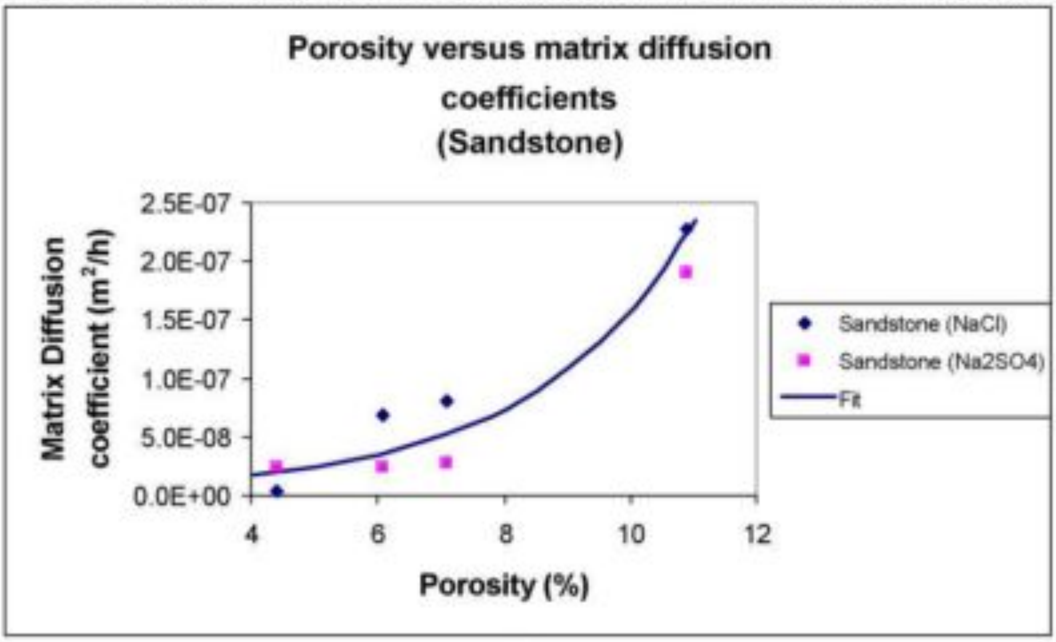
The same matrix diffusion experiment was performed on various sandstones, shales and one quartzite. The cores were all approximately 5 mm thick. A solute of 100000 mg/L NaCl or 100000 mg/L Na_2SO_4 was added to cell A, while deionized water was added to cell B. The results of the experiments can be seen in Figure 2-10. The calculated matrix diffusion coefficients are documented in Table 2.1.

Table 2.1: Matrix diffusion coefficients

Formation	Porosity (%)	D (m^2/h) NaCl	D (m^2/h) Na_2SO_4
Sandstone (coarse grained)	10.9	2.28×10^{-7}	1.89×10^{-7}
*Sandstone (medium grained)	7.1	8.02×10^{-8}	2.68×10^{-8}
Sandstone (medium grained)	6.1	6.82×10^{-8}	2.41×10^{-8}
Sandstone (fine grained)	4.4	3.34×10^{-9}	2.41×10^{-8}
Shale	1.12	1.88×10^{-7}	1.87×10^{-8}
Shale	0.92	1.78×10^{-7}	2.19×10^{-8}
Shale	0.83	7.59×10^{-7}	-
Quartzite	0.19	1.80×10^{-7}	-

*Sandstone with fracturing present

Figure 2-10 (a). Relationship between porosity and matrix diffusion coefficients for



sandstones

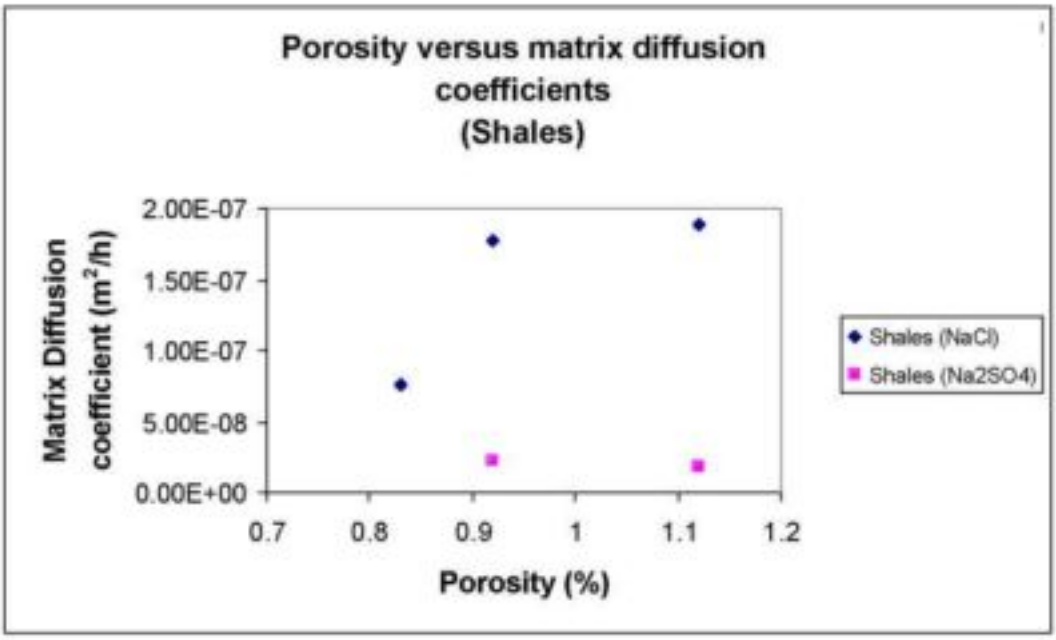


Figure 2-10 (b). Relationship between porosity and matrix diffusion coefficients for shales

From Figure 2-10(a), a relationship between porosity and matrix diffusion coefficients for sandstones can be determined as:

$$D = 3.2\text{x}10^{-9}\ \exp^{0.39n_e}$$

where D is the matrix diffusion coefficient calculated in m^2/h and n_e is the porosity. This equation is based on very little data and numerous experiments will have to be conducted to validate this result. No relationship between porosity and the matrix diffusion coefficient can be determined from Figure 2-10 (b) as there is insufficient data. The matrix diffusion coefficients do not show the same trends as those in the sandstones. This may be a result of either interactions between the shales and the concentrations of NaCl and Na_2SO_4 and/or secondary porosity. Due to difficulties in obtaining quartzite core only one type of quartzite was used in the experiments. It is interesting to note, however, that even though the porosity of the quartzite is low, the diffusion coefficient is relatively high. This is most probably due to the fact that there is very little interaction between the quartzite and NaCl or Na_2SO_4.

Matrix diffusion coefficients for various concentrations

It is sometimes easier to understand diffusion coefficients when expressing them in terms of mass of contaminant that passes from the fracture into the matrix. Once the matrix diffusion coefficient has been determined, Equation 2.3 can be used to calculate the mass passing through the core in an hour. To demonstrate this, three matrix diffusion experiments were performed on the same sandstone (with only primary porosity) using different concentrations. The core radii were 30 mm in all cases and the thickness was 5 mm. The matrix diffusion coefficients, together with the concentrations in cell A and the transfer rate of NaCl, are listed in Table 2.2.

Table 2.2: Comparison of matrix diffusion values for various concentrations

Initial NaCl concentration in cell A (g/10 ml deionized water)	**Matrix diffusion coefficient (m^2/h)**	**NaCl per area core (mg/h)**
0.5	3.34×10^{-9}	0.1
1.0	2.12×10^{-8}	1.2
2.0	4.99×10^{-8}	5.6

An indication of the quantity of solute that can diffuse into a rock matrix, the values in Table 2.2 are used to calculate the amount of NaCl that can diffuse via fractures in varying areas (Table 2.3).

Table 2.3: Amount of NaCl that can diffuse into a rock matrix (with primary porosity) using values listed in Table 2.2

Areal extent of fracture (m x m)	**Amount of NaCl that can diffuse into matrix (g/h) when D is**		
	3.34×10^{-9}	**2.12×10^{-8}**	**4.99×10^{-8}**
1 x 1	0.0071	0.085	0.4
10 x 10	0.707	9	40
50 x 50	18	212	989
100 x 100	71	848	3958

When plotting the values listed in Table 2.2 one can see an "almost" linear relationship between the initial NaCl concentrations in cell A and the matrix diffusion coefficients (Figure 2-11 (a)). A similar relationship can be seen when comparing the concentration in cell A and NaCl movement through the core (Figure 2-11(b)).

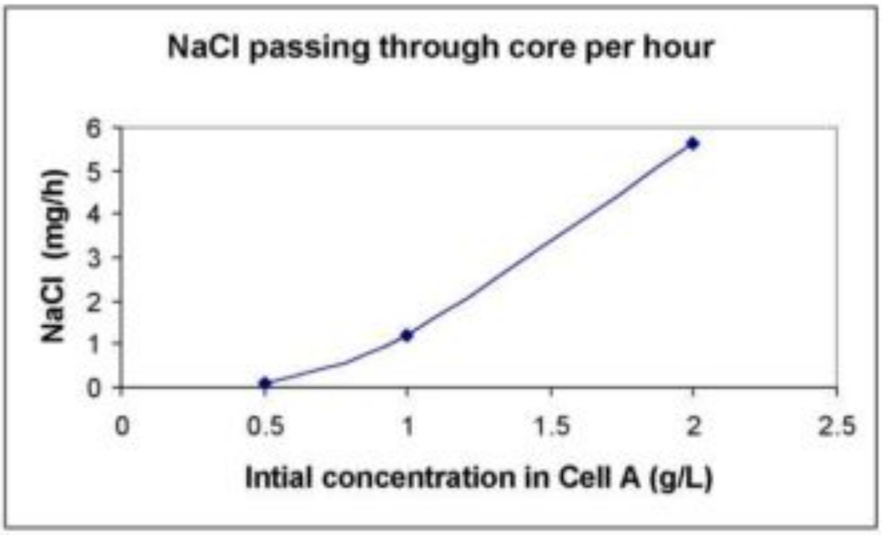


Figure 2-11(a). Amount of NaCl passing through sandstone core per hour for various initial NaCl concentrations in cell A.

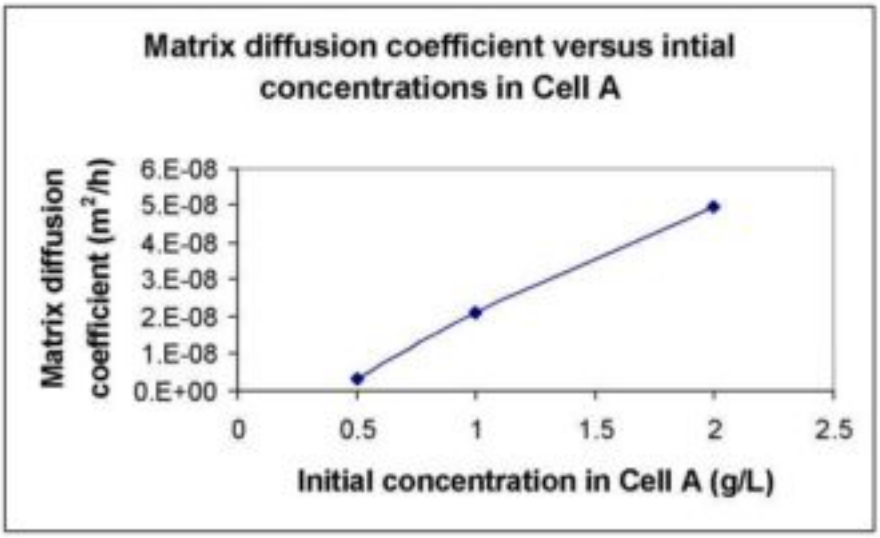


Figure 2-11(b). Increase in matrix diffusion coefficients as initial concentrations of NaCl increase in cell A.

Determining the difference between matrix diffusion in the horizontal and vertical directions

All afore-mentioned matrix diffusion coefficients have been determined for horizontal slices and, therefore, in the vertical direction (refer to Figure 2-12). However, the question remains as to whether or not there is a difference between horizontal and vertical matrix diffusion values. A number of experiments on sandstones were conducted to study this aspect.

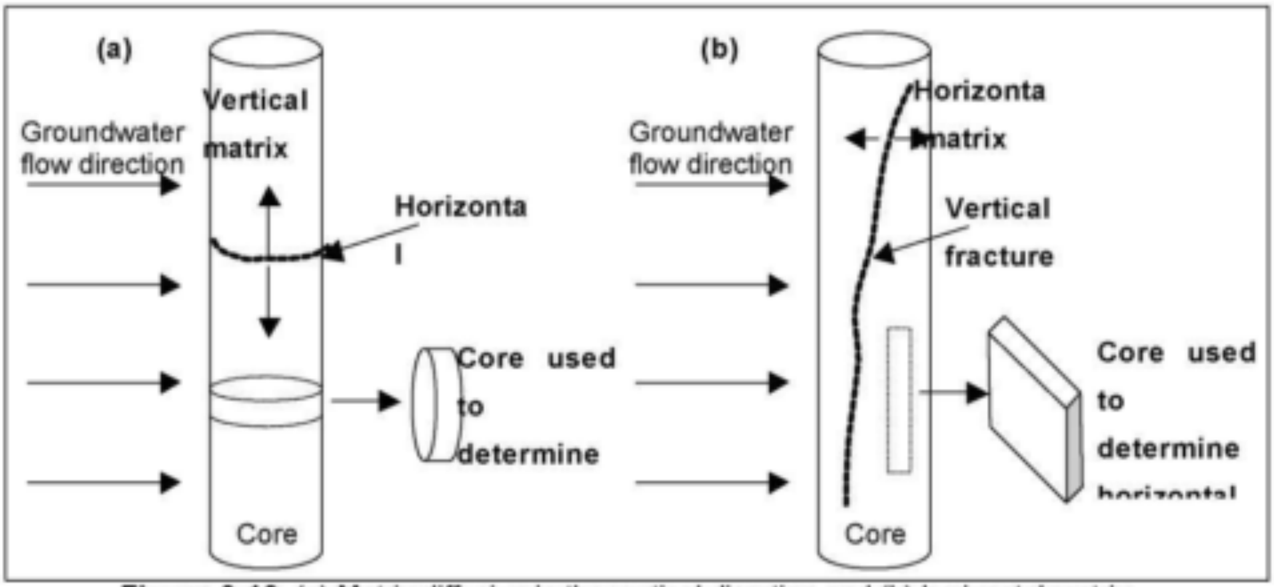


Figure 2-12. (a) Matrix diffusion in the vertical direction and (b) horizontal matrix diffusion

The results of these experiments are listed in Table 2.4.

Table 2.4: Horizontal versus vertical matrix diffusion coefficients

Horizontal coefficient	Vertical coefficient
9.95×10^{-8}	3.31×10^{-8}
8.24×10^{-7}	2.74×10^{-7}
8.92×10^{-7}	2.12×10^{-8}

In all cases the matrix diffusion coefficients in the horizontal direction are greater than those in the vertical direction. The degree of difference varies from sandstone to sandstone and is most probably dependent on factors such as the fracture characteristics and mineral composition of the sandstone.

Discussion

The experiments discussed in this section provide an indication of the impacts of matrix diffusion. Table 2.3 provides an idea of the amount of pollution that can diffuse from a fracture acting as a conduit for pollutants into the adjacent rock matrix. It is important to note that the results discussed in this section are based on a few experiments and must be verified. Unfortunately, the duration of each experiment exceeds one month. Combined with many problems such as cells that leaked and EC probes which did not read correct values, especially in cell A where the EC values were extremely high, it was not possible to conduct more experiments within the time frame of this study.

2.3.3 Tracer Tests *(Summarised from Van Tonder et al., 2001)*

2.3.3.1 General

For the investigation of risk assessment and remediation of groundwater contamination, it is important to estimate transport parameters such as groundwater velocity, effective (or kinematic) porosity and dispersion. For high confidence results, these parameters have to be analysed from field tests, known as tracer tests. As tracer tests under natural conditions, with several observation boreholes, require much time and are costly, different single-well and dual-well tracer tests were explored, two of which will be discussed in this section.

For more information concerning tracer test analysis and related software refer to Appendix A.

2.3.3.2 Single Well Injection-withdrawal Test

To conduct a single well injection-withdrawal test, a tracer is introduced to the standing water column of the test borehole and allowed to drift away, under natural gradient, from the borehole. The test borehole is pumped until the tracer plume is retrieved. Groundwater flow velocity is then calculated based on the amount of pumping needed to recover the tracer.

$$v = \frac{\left(\dfrac{Qt_p}{n_e b^{3-n} \beta_n}\right)^{1/n}}{t_d} \quad \text{and} \quad \beta_n = \frac{\pi^{n/2}}{\dfrac{1}{2} n \Gamma\left(\dfrac{n}{2}\right)}$$

where

Q = Pumping rate during recovery of tracer

n_e = Effective porosity

t_p = Time elapsed from start of pumping until the centre of mass of the tracer is recovered

t_d = Time elapsed from the injection of tracer until the centre of mass of the tracer is recovered

The effective porosity can be calculated:

$$n_e = \left[\frac{\beta_n b^{3-n}\left(K\dfrac{dh}{dl}t_d\right)^n}{Qt_p}\right]^{\frac{1}{n-1}}$$

where

dh/dl = Hydraulic gradient

2.3.3.3 Radial Convergent Test

Pumping a borehole until steady state conditions are reached creates a radial convergent flow field. A tracer is then quickly introduced into an injection borehole in the vicinity of the pumping borehole in such a way that minimum disturbance of the flow field is caused, while the tracer breakthrough curve is monitored at the pumping borehole. Analyses of the resulting breakthrough curves yield estimates of the effective porosity, aquifer dispersivity and groundwater velocity. The convergent test is attractive because it is theoretically possible to recover the tracer from the aquifer. Furthermore, it more closely represents reality as groundwater pollution often occurs in the vicinity of pumping boreholes where radial flow fields are present. The approximate solution for converging radial flow with a pulse injection is given by:

$$C(r,t) = \frac{\Delta M}{2Q\sqrt{\pi \alpha_L v t^3}} \exp\left[-\frac{(r - vt)^2}{4D_L t}\right]$$

where

ΔM	=	Injected mass of tracer per unit section
α_L	=	Longitudinal dispersivity
D_L	=	Longitudinal dispersion coefficient
v	=	v_f; groundwater velocity under forced gradient
Q	=	Pumping rate of the borehole
r	=	Distance between the two boreholes

The flow velocity under forced gradient v_f and dispersivity can be estimated by fitting the equation to the data of the breakthrough curve. The effective porosity can then be estimated using the following equation:

$$n_e = \frac{Q}{vA}$$

where A is the through flow area:

$$A = \frac{2\pi^{n/2} r^{n-1} b^{3-n}}{\Gamma\left(\frac{n}{2}\right)}$$

CHAPTER 3

The Decision Tool

3.1 BACKGROUND

During the last few years, the thinking concerning the management and protection of water resources has changed radically in South Africa. A *resource directed measures* (RDM) team was initiated to ensure that certain aspects of the National Water Act would be implemented. The task of this team included:

- To devise a system of consistent rules to guide decision-making about water resources on a national basis.
- The national system should allow transparency, accountability and long-term goal-setting to be incorporated into water resources management.
- Water resources that need to be improved can then be identified and the necessary control measures can be implemented to meet the requirements (MacKay, 1998).

Depending on the importance and sensitivity of the groundwater resource, there are various levels of determinations:

- *Desktop estimate* – a short planning estimation, with very low confidence attached to the results.
- *Rapid determination* – an extension of the desktop study. Low levels of confidence are attached to the results.
- *Intermediate determination* – this includes specialist field studies. Medium levels of confidence are attached to the results.
- *Comprehensive determination* – a relatively high confidence is attached to this determination and includes extensive field data collection by specialists.

To align the decision tool with South African reserve determinations, a tiered approach was followed. The first tier is a rapid assessment in which only existing data is required and it produces low confidence results. It is intended to give the assessor a guideline of the risks and cost implications involved. The next tier is an intermediate assessment. The first step in the intermediate assessment is to collect all relevant data. Data requirements include recharge values, aquifer and contaminant parameters, as well as health and ecological information. Most of the general information will be obtained from the database included in the DT software, but it is sometimes necessary to have site-specific data. The confidence attached to these results is low to medium. Finally a comprehensive assessment requires extensive field investigations and specialist studies. Once all necessary data has been collected, it will be analysed. The confidence attached to the comprehensive assessment should be medium to high.

3.2 THE FUZZY LOGIC BASED SYSTEM *(Summarised from Van der Werf and Zimmer, 1997)*

Fuzzy logic is a superset of conventional (Boolean) logic that has been extended to handle the concept of partial truth; truth values between *completely true* and *completely false*. It was proposed by Zadeh (1965) to deal with uncertainty.

In classical set theory, an element is either in a set or it is not. For example, if a subset A consists of pesticides with a maximum field half-life of 20 days, a particular pesticide can be classified as a member or not a member of a subset. If, however, A is defined to be the subset of 'non-persistent' pesticides, then it is more difficult to determine if a specific pesticide is in the subset. If one decides that only pesticides with a maximum field half-life of 20 days are in the subset, then a pesticide with a 21-day half-life cannot be classified as non-persistent even though it is *almost* non-persistent. The use of fuzzy set theory is particularly compelling because available values for field half-life and several other relevant variables are imprecise and/or uncertain.

Fuzzy set theory addresses this type of problem by allowing one to define the degree of membership of an element in a set by means of a *membership function*. For classical sets, the membership function only takes 2 values: 0 (non-membership) and 1 (membership). In fuzzy sets the membership function can take any value from the interval [0,1]. The value 0 represents complete non-membership, the value 1 represents complete membership and the values in between are used to represent partial membership (transitional zone).

For input variables two fuzzy subsets F (favourable) and U (unfavourable) are defined. The membership functions are based on *available data* or *expert knowledge*. Many membership functions are sine shaped in the transitional interval. For example consider the leachability of pesticides. If the groundwater ubiquity score is greater than 2.8 then a membership of 1 is assigned and the values are totally unfavourable. Pesticides are classified as non-leachers if the ubiquity score is less than 1.8. A membship value of 0 is given for the fuzzy subset U and a membership value of 1 is given for the fuzzy subset F. The class of borderline components between 1.8 and 2.8 falls within the transition zone in which the membership value for F decreases from 1 (ubiquity score = 1.8) to 0 (ubiquity score = 2.8) and the membership value for U increases from 0 to 1. The functions characterising F and U are therefore complementary (Figure 3-1).

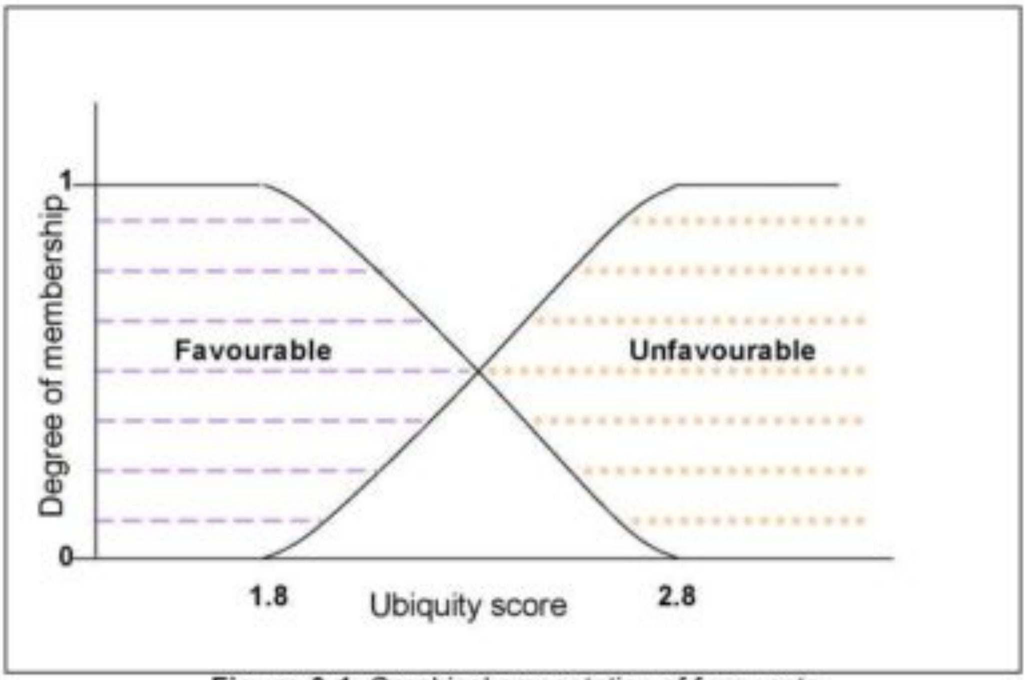


Figure 3-1. Graphical presentation of fuzzy sets

For each component of the tool a set of decision rules is formulated attributing values between 0 and 1 to an output variable, according to the membership of its input variables to the fuzzy subsets F and U. Sugeno's (1985) inference method is used to compute the decision rules and risk. The decision rules take the form:

If x_1 is A_{11} and x_2 is A_{12} then y is B_1

If x_1 is A_{21} and x_2 is A_{22} then y is B_2

Where x_j (j = 1, 2) is an input variable (eg half-life), y is an output variable (eg value of the component). A_{ij} is a fuzzy set and B_i is a number known as the conclusion of the rule.

Let x_1' and x_2' be the values taken by x_1 and x_2, and $A_{ij}(x_j')$ the membership value of x_j' to the fuzzy set A_{ij} (given by the membership function that defines A_{ij}). The truth values are then defined as:

$$W_1 = \min(A_{11}(x_1'),A_{12}(x_2'))$$

$$W_2 = \min(A_{21}(x_1'),A_{22}(x_2'))$$

Where min is the 'minimum value of'. The first rule infers W_1B_1 and the second one infers W_2B_2. The final risk can then be defined as

$$\text{Risk} = (W_1B_1 + W_2B_2)/(W_1 + W_2) \qquad (3.1)$$

The above explanation is easier to understand when explained by means of an example. Assume (only to illustrate the approach) that the risk of groundwater contamination from a pesticide depends on two input variables: rate and duration of application. For both input variables membership to fuzzy sets F and U have to be defined. Assume that the experts say that a low rate of application and a short impact period are favourable, whereas a high rate and duration of application are unfavourable. For the rate of application complete membership to the fuzzy set F is if the rate of application < 0.001 kgha^{-1} and complete membership to the fuzzy set U if the rate of application > 2 kgha^{-1}; for duration complete membership to F is if duration of application < 1 day and complete membership to U if duration of application > 120 days (Figure 3-2).

In this example, there are 2 input variables and two fuzzy subsets for each input variable therefore 4 situations may occur, as reflected in the decision table (Table 3.1). These rules reflect expert knowledge and/or expert judgement. The first line of the table reads as *if the rate of application is favourable and the duration of application is favourable then the rule conclusion is 0.* As can be seen from Table 3.1, when both input variables are F, the rule conclusion is 0, when both input variables are U the rule conclusion is 1. When one input variable is F and the other is U the rule conclusion is 0.5.

Table 3.1: Summary of decision rules describing the effect of input variables rate of application and duration of application on risk of contamination

Rate of application	Field half-life	Conclusion
F	F	0.0
F	U	0.5
U	F	0.5
U	U	1.0

Once the membership functions and decision rules have been defined, the output for the risk of contamination from the pesticide can be defined. Assume that atrazine is applied at 1.5 kgha^{-1} for 60 days. The membership functions defined allow one to calculate the degree of membership for this pesticide (Figure 3-2).

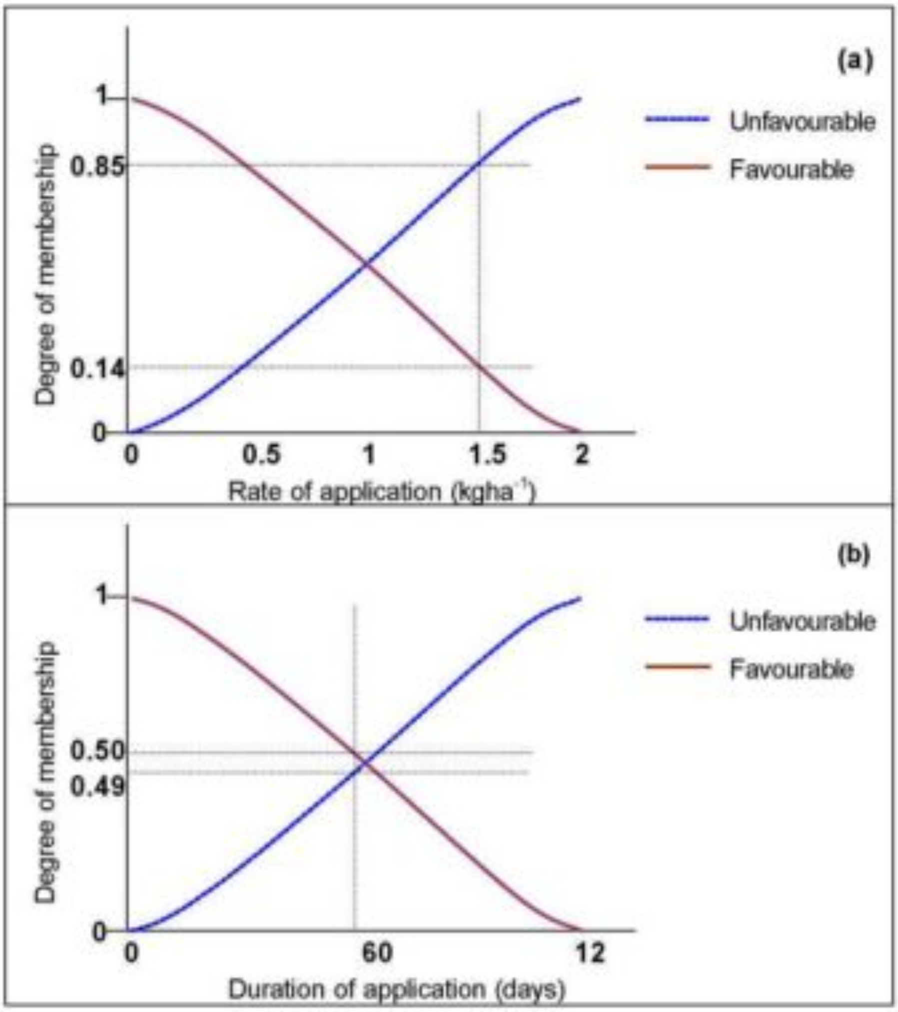


Figure 3-2. Membership to the fuzzy sets favourable and unfavourable for atrazine (a) rate of application and (b) duration of application

According to Sugeno's (1985) inference method, the truth value of a decision rule can be defined as the smallest of the truth values. The value for the final risk is calculated as the average of the conclusions of the decision rules, weighted by their truth according to Equation 3.1:

$$\text{Risk} = \frac{(0 \times 0.147 + 0.5 \times 0.147 + 0.5 \times 0.506 + 1 \times 0.494)}{(0.147 + 0.147 + 0.506 + 0.494)} = 0.634$$

This implies that the risk of possible contamination of groundwater from pesticides (being applied under the above-mentioned conditions) is 63.4 % .

3.3 STRUCTURE OF THE DECISION TOOL

For each of the tiers the following risk assessments can be performed:

- A groundwater risk assessment can be defined as the probability of an adverse effect or effects on the quantity and/or quality of groundwater associated with measured or predicted hazards (for example chemical spills).

- A groundwater health risk assessment can be defined as a qualitative or quantitative process to characterise the probability of adverse health effects associated with measured or predicted levels of hazardous agents in groundwater. The health risk assessment is divided into a carcinogenic, a radiogenic, a non-carcinogenic and a microbial assessment.
- Ecological risks of interest differ qualitatively between different stresses, ecosystem types, and locations. A groundwater ecological risk assessment quantifies the impacts of groundwater quantity and quality on ecosystems. However, according to the reserve calculations (MacKay, 1998) only aquatic ecosystems need to be considered. Therefore emphasis will be placed on these ecosystems.

Information from both the groundwater sustainability and contaminant risk assessments can be used as input data for health and ecological risk assessments.

Once the desired risk assessments have been completed, a cost-benefit-risk analysis can be used to aid in decision-making regarding the management and remediation of a groundwater resource. A cost-benefit-risk analysis is defined as a set of procedures used for defining, comparing and measuring benefits and costs, which originate from either an investment or the operation of a service. The cost-benefit-risk analysis is a flexible and adaptable method; however, the assessor should keep the suitability of a cost-benefit-risk analysis for different types of projects in mind before an attempt is made to perform the analysis.

The method used to determine a cost-benefit-risk analysis is similar to that documented by Rosen and LeGrand (1997). Monetary risk is defined as $R = PC$ where P is the risk of failure and C represents the economical consequences associated with the failure expressed in monetary terms. To choose between the different alternatives an objective function is set up including the benefits, costs and risks of a project.

Since the early 1980's geohydrologists and engineers have developed a number of techniques for protecting groundwater. The government places such great emphasis on protection of water resources that a whole chapter of the National Water Act (Act No. 36, 1998) is dedicated to this topic. Protection is divided into two categories: measures to prevent failure and pollution of water resources and measures to remedy the effects of pollution of water resources.

On completion of the different aspects of the DT, a report including the input data and results of the risk assessments and cost-benefit-risk analysis will be generated. Depending on the user, prevention measures can be included. Unfortunately, no in-depth study has been completed for remediation options but the user will be able to do a search on various options. The results of the search can be included in the report.

The information obtained from the DT can be used for risk management. Risk management is defined as the process of identifying, evaluating, selecting and implementing actions to reduce risks.

Each of the components of the DT will be discussed in more detail in the following chapters.

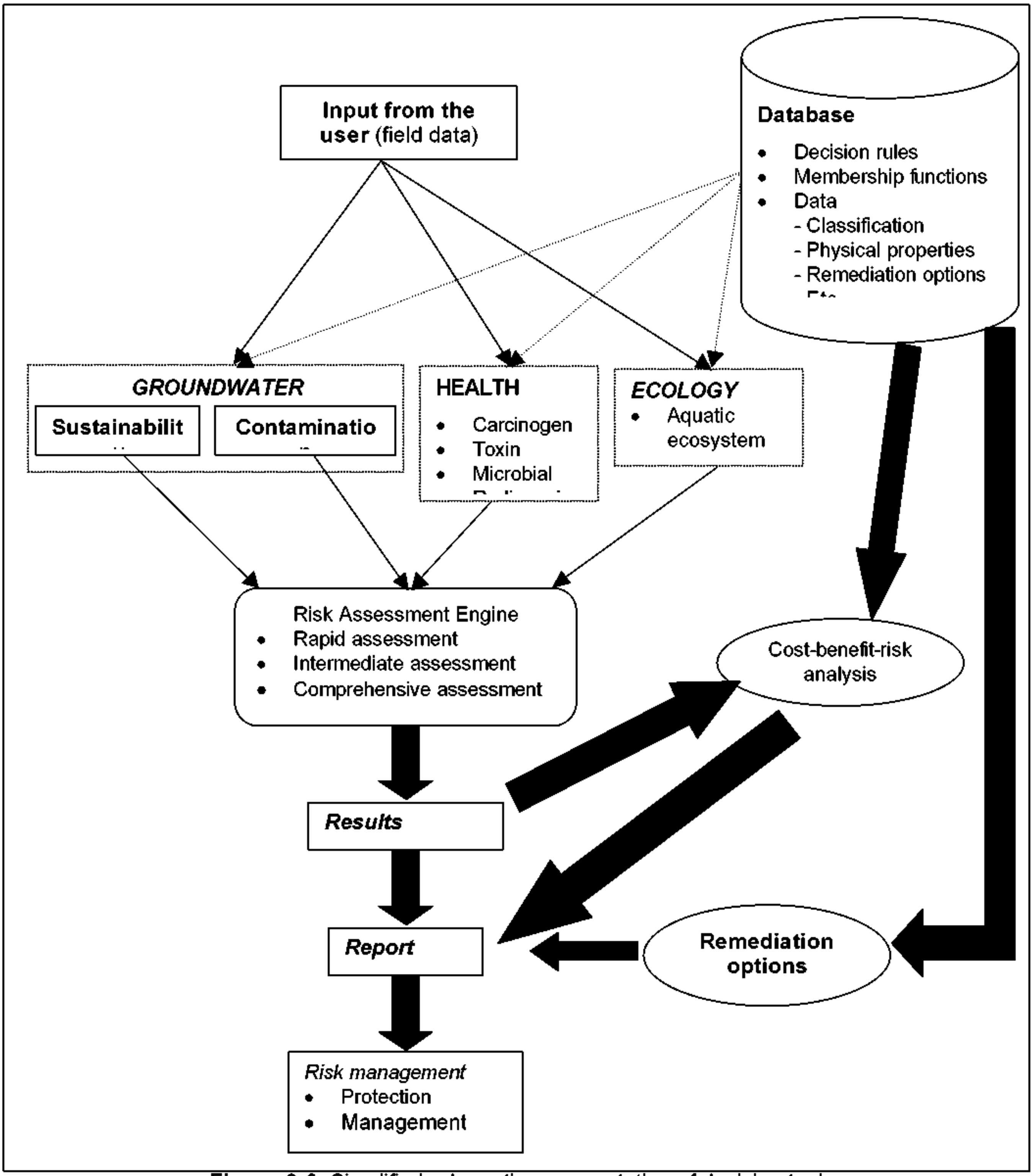
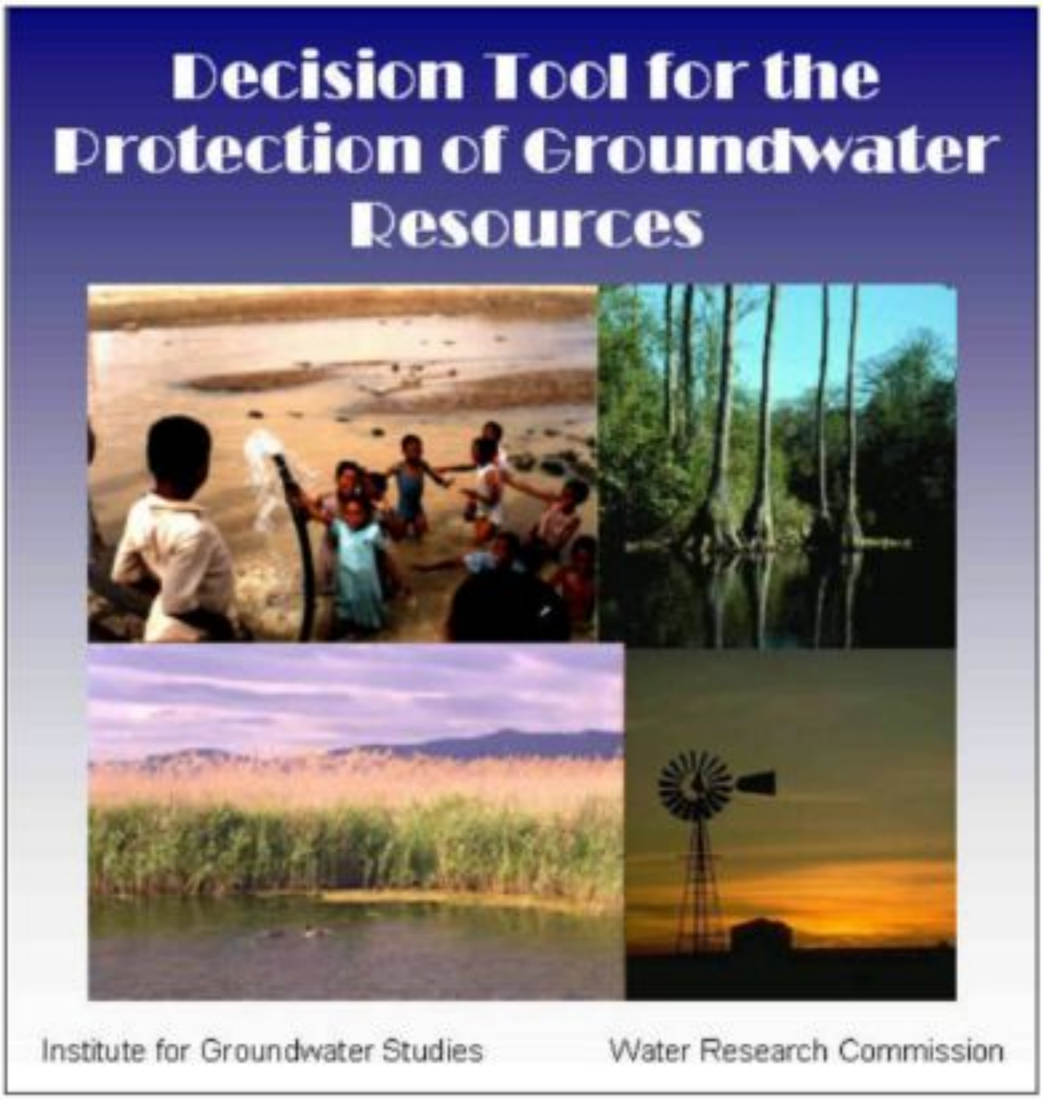


Figure 3-3. Simplified schematic representation of decision tool

3.4 DECISION TOOL SOFTWARE

The DT software was developed by SR Dennis using Borland C++ Builder Pro. The DT is linked to an MS Access Database in which all data, decision rules and membership function data are stored. The start-up screen for the DT is shown in Figure 3-4.

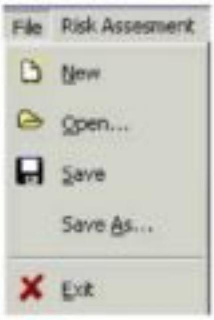
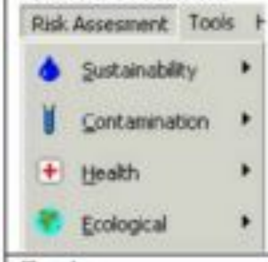
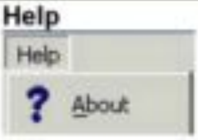
Figure 3-4. Start-up screen for decision tool



The functions on the main menu of the DT are summarised in Table 3.2.

Table 3.2: Functions on main menu of decision tool

Main menu	Function

File 	Contains usual software functions such as "new", "open", "save", "save as" and "exit". "Open", "save" and "save as" allow the user to call up an existing project file without having to start from the beginning every time.
Risk Assessment 	The risk assessment menu contains all risks assessments that can be performed by the DT. Each of the assessments is further divided into a rapid, intermediate and comprehensive assessment.
Tools 	Under the "tools" menu there is the remediation and prevention option, the cost-benefit-risk analysis and the report generator that takes all data into account. The database composer gives the user access to the database which contains decision rules, membership functions, data etc.
Help 	The help function provides information concerning the people and institutions involved in developing the DT. However, there are other help functions available in various sections of the DT.

Once the start-up screen has disappeared, the main screen of the DT will appear (Figure 3-5). Here the user can decide which risk assessment he/she would like to do. By default the rapid assessment for sustainability appears. However the user can move to any of the other assessments by selecting the respective tab. Each of the assessments together with the cost-benefit-risk analysis, prevention and remediation, and the generation of reports will be discussed in more detail in the examples in Chapters 4 – 9. The only aspect of the DT still to be mentioned here is the database. The only way to enter the database is by means of a password. Once the access code has been entered the main database screen appears. The structure of the database is similar to Windows Explorer. For example, consider the sustainability risk assessment. All information needed for this risk assessment is stored under sustainability (Figure 3-6). Similarly information for the other risk assessments is stored in the database and can be accessed and changed here.

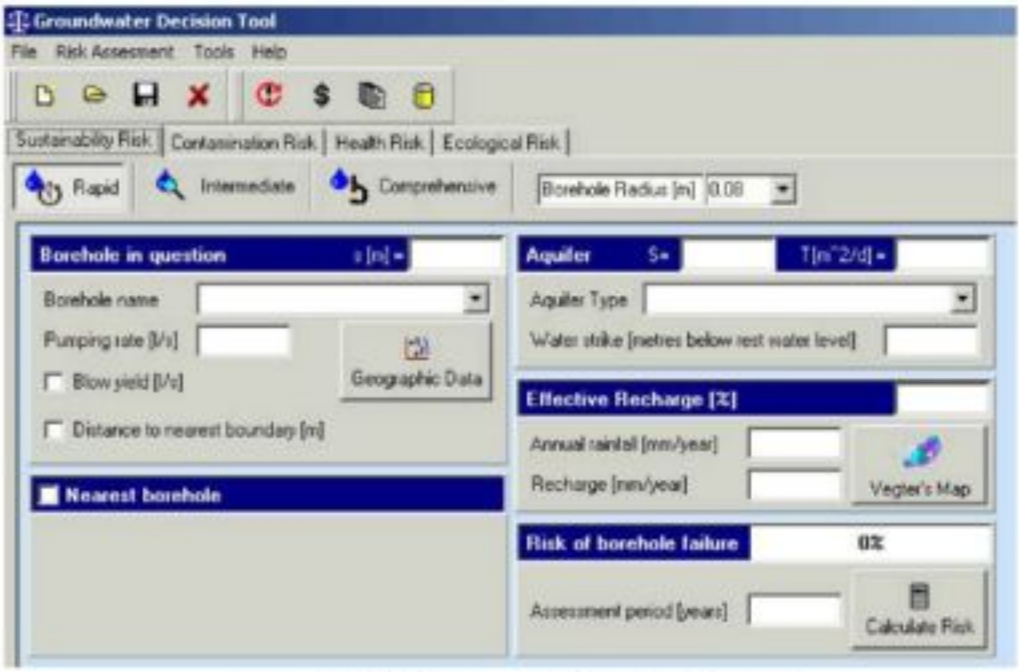


Figure 3-5. Main screen of the decision tool

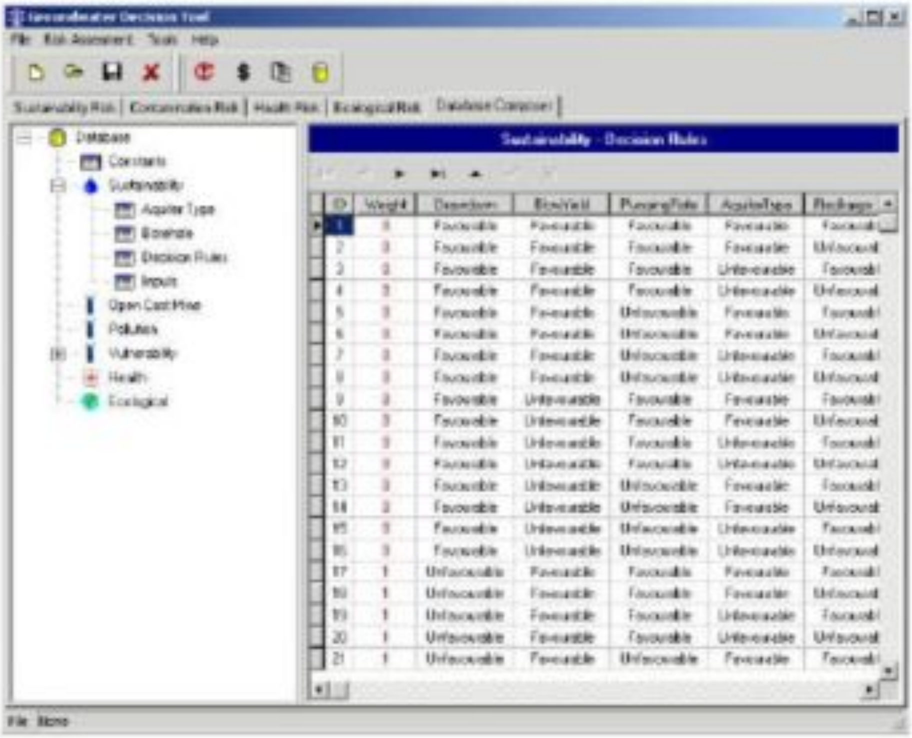


Figure 3-6. Database of the decision tool

CHAPTER 4

Groundwater Sustainable Risk Assessment

4.1 PREAMBLE

Braune (2000) stated: *Groundwater is particularly vulnerable to poor management. This is because of its 'invisible' nature, the often delay before over-exploitation manifests itself and the limited self-purification. Once groundwater becomes polluted, it is difficult, if not impossible, to rehabilitate. Unfortunately groundwater resources, both in quality and yield, are put at risk by a wide range of human activities. These should be managed to ensure the sustainable utilisation of the resource. . . . To avoid unnecessary risks to groundwater resources, requires knowledge-based management. However, obtaining such knowledge in the case of groundwater is an incremental process necessitating a precautionary approach to all groundwater management decisions. Strategies and actions should be pro-active, planned and preventative, wherever possible, rather than reactive*.

As a result, pro-active methodologies must therefore be developed in order to ensure the protection of this valuable resource. One such methodology uses risk assessments based on fuzzy logic to determine the potential failure of a groundwater resource, taking into account both the quantity and quality of the resource. The methodologies capture the knowledge of experts in the field of groundwater sustainability and contamination. This chapter will focus on the groundwater sustainablity or quantity risk assessment. Chapter 5 will deal with quality or contamination risk assessment.

4.2 GROUNDWATER SUSTAINABLE RISK ASSESSMENT

4.2.1 General

There are many definitions for groundwater sustainability. Sharp (1998), for example, defined the sustainable yield of groundwater as the minimisation of potential negative effects on an aquifer so that it can be utilised at an acceptable range of levels for a very long period of time. Merrick (2000) stated that sustainable yield is that proportion of the long-term annual recharge, which can be abstracted each year without causing unacceptable impacts on groundwater users or the environment. Van Tonder's (2001) definition of sustainable yield is that it is the safe amount of water that can be abstracted from a borehole for a long time (usually 1 or 2 years), without the water level reaching the position of the pump or the main water strike.

An increasing number of boreholes in Southern Africa have dried up during the past years in spite of comparable constant hydrologic conditions (for example Petrusburg). A new investigation of reliable estimates for the sustainable yield of the boreholes was therefore required. Overestimation of the borehole yield was due to the application of improper extrapolation of drawdown curves, which ignored barrier boundaries and neglected parameter uncertainties arising from the imperfect knowledge of the effective aquifer properties (Van Tonder *et al.*, 1999).

A groundwater quantity risk assessment has therefore been designed to determine the risks of failure when abstracting from an aquifer. The authors felt it important to consider not only recharge, but also other important factors to ensure that the water level does not reach the main water strike or pump position. These factors include:

- Blow yield, which can be used to determine an estimate for the sustainable borehole yield.
- Recharge, which is an important factor according to the definitions of sustainable yield.
- Water strike/depth of main fracture which determines the amount of drawdown possible in a borehole. According to Van Tonder's (2001) definition of sustainable yield, it is important not to abstract a quantity of water such that the water level reaches the water strike or pump. The drawdown is calculated taking into account the influence of other abstraction boreholes and boundaries.
- The aquifer type will determine the amount of water that can be released from storage.
- The period for which the users would like to abstract is important. Calculations show (Van Tonder and Dennis, 2000), the longer the period of abstraction, the larger and deeper the cone of influence.
- Slug and pumping tests are used to determine transmissivity or hydraulic conductivity values.

The methodology to determine the risk of borehole or aquifer failure due to over abstraction is summarised in Figure 4-1.

4.2.2 Data Requirements

One of the major differences between the three tiers of assessments is the quality of data used for each assessment. This can be seen from the data requirements listed in Table 4.1. Maps and databases (containing data from literature) are used in the rapid assessment. Data from at least one field investigation is included in the intermediate assessment thereby improving confidence in the results. The comprehensive assessment requires a full set of field data to attach a medium to high confidence to the results.

4.2.3 Assumptions and Limitations

The assumptions and limitations for the various assessments differ and will therefore be discussed separately.

Rapid Assessment

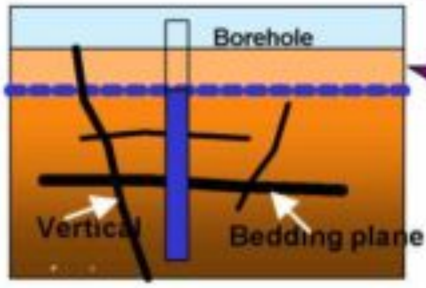
- Only the influence of the closest no-flow boundary and abstraction borehole is taken into account.
- Literature, maps, rules of thumb and simple calculations are used to determine the membership functions and decision rules.
- The Cooper-Jacob method is used to determine the drawdown at the borehole under investigation. It is also used to determine the drawdown at the borehole under investigation as a result of the closest abstraction borehole.
- When determining the drawdown in a borehole under investigation, the actual radius of the borehole is used and not the effective radius.
- The risk assessment can only be conducted on one borehole.

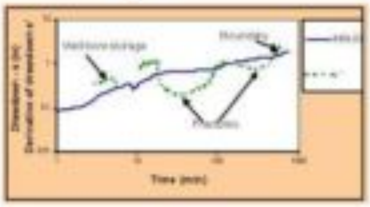
Groundwater sustainable risk assessment

Problem formulation

Hazard identification establishes whether abstraction at a given rate will negatively impact the aquifer.

Questions to be asked: What is the problem? Why is it a problem? Are the effects likely to appear in the near future, or in future generations? How urgent is the need for action?





Analysis

Establishes the tendency or likelihood for abstraction to have negative effects on the groundwater levels. Establishes risks of potential impacts taking into account the properties of the aquifer.

Risk characterisation

Provides an indication of the incidence of possible impacts taking into account properties of the aquifer and the abstraction boreholes. Risk characterisation should include information that is useful to both stakeholders and risk managers. *Questions to be asked:* What is the nature and likelihood of the risk? How severe are the anticipated adverse effects? Are the effects reversible? What scientific evidence supports the conclusions about the risk? How strong is the evidence? What is uncertain about the nature or magnitude of the risk?

Figure 4-1. Summary of methodology for sustainable risk assessment

Table 4.1: Data required for sustainable risk assessment and potential data sources

Data required	Potential data sources		
	Rapid	**Intermediate**	**Comprehensive**
Blow yield	User/guesstimated by DT	-	-
Recharge			
• Recharge	User/recharge map[1]	User/recharge map[1] OR	User OR
• Rainfall	Weather bureau/user	Weather bureau/user	Weather bureau/user
• Chloride values in groundwater and rainfall	-	Laboratory analyses/database	Laboratory analyses OR
• Water levels			Field data
Other abstraction boreholes[2]			
• Distance between investigated borehole & other abstraction boreholes	User OR	User OR	User OR
• Positions of boreholes	User	User	User
• Abstraction rates	User	User	User
Aquifer information			
• Type/storativity	User[3] (drop down menu in DT)	User[3] (drop down menu in DT)	User[3] (drop down menu in DT)
• Transmissivity	User		-
○ Slug test data	-	Field data OR	-
○ Pumping test	-	Field data	Field data
Water strike/fracture	User	User/pumping test data/borehole log	User/pumping test data/borehole log
Aquifer boundaries			
• Distance to closest no-flow boundary	User OR	-	-
• Positions of aquifer no flow boundaries	User (*.bnd file) can be determined from field investigations, geological or topographic maps	User (*.bnd file) can be determined from field investigations, geological or topographic maps	User (*.bnd file) can be determined from field investigations, geological or topographic maps
Period for which the borehole under investigation is to be pumped.	User	User	User

[1]Map included in Decision Tool (See Appendix B, Section B1).

[2]Only closest abstraction borehole to borehole under consideration taken into account for the rapid assessment.

[3]Data used in the drop down menu is documented in Appendix B, Section B2.

- Data not needed for assessment.

GENERAL NOTE: Manual override of calculated values is possible.

Intermediate Assessment

- Five different boundary configurations are taken into account, namely:
    - No boundary
    - Single barrier boundary
    - Two barrier boundaries intersecting at 90°
    - Two parallel boundaries
    - Closed square boundaries

  Refer to Appendix B (Section B3) for more information concerning the boundary conditions.
- The Cooper-Jacob method is used to determine the drawdown at the borehole under investigation. It is also used to determine the drawdown at the borehole under investigation as a result of other abstraction boreholes in the area.
- When determining the drawdown in a borehole, the actual radius of the borehole is used and not the effective radius.
- The risk assessment can be conducted on numerous boreholes, but the highest calculated risk will be taken as the final risk of the wellfield.

Comprehensive Assessment

- The five different types of boundaries listed under the intermediate assessment are taken into account.
- The influence of a fracture is included using the Barker's GRF-model and the non-integer flow dimension (refer to Chapter 2).
- When determining the drawdown in the borehole under consideration, the actual radius of the borehole is used and not the effective radius.
- The risk assessment can be conducted on numerous boreholes, but the highest calculated risk will be taken as the final risk of the wellfield.
- The user must use the Hurst method to determine n and the GRF-model to determine the drawdowns. This allows the user to take the fracture nature of the aquifer into account. If working with a porous aquifer n should be 2 and the GRF-model will give the same results as the Theis model. If, however, the user would prefer using the Cooper-Jacob method, then the intermediate assessment can be used.

4.2.4 Methodology

The methodologies (Table 4.2) differ slightly when calculating the data required for the membership functions and therefore these calculations will be discussed separately for each of the assessments. The calculations vary from elementary calculations for the rapid assessment to complex sophisticated calculations for the comprehensive assessment.

Table 4.2: Methodologies for calculating a sustainable assessment

Data required for various calculations	Calculations		
	Rapid assessment	**Intermediate assessment**	**Comprehensive assessment**
Blow yield: A common rule-of-thumb in groundwater circles indicates that the rate at which a borehole can be pumped is approximately 20% of the blow yield for 24 hr/day or 60% for 8 hr/d. Therefore, blow yield can be used as a first estimate of the sustainable yield of a borehole.	If the blow yield of the borehole is not known, it can be estimated using the following equation: blow yield = 1.7 x current pumping rate.	If the blow yield of the borehole is not known, it can be estimated using the following equation: blow yield = 1.7 x current pumping rate.	If the blow yield of the borehole is not known, it can be estimated using the following equation: blow yield = 1.7 x current pumping rate.

Blow yield: Is used to determine transmissivity for the rapid assessment. Slug test/pumping test data: The data from these tests are used to calculate the transmissivity/hydraulic conductivity values in the intermediate and comprehensive assessments.	The blow yield is also used to determine the transmissivity of the aquifer using the following rule-of-thumb: T (m/d) = 10 x 0.6 x blow yield (L/s).	Slug test data are used to calculate the yield of a borehole Q (L/h) = $117155.08t^{-0.824}$ Where t is the recession time of the slug test in seconds (Vivier et al., 1995), the transmissivity is then calculated using the rule-of-thumb: T (m/d) = 10 x Q (L/s) OR Use the data of one pumping test to obtain a value for T using the Cooper-Jacob method (Kruseman and De Ridder, 1991) OR Logan's formula (Misstear, 1991) can be used to determine the transmissivity: $T \approx 1.22\frac{Q}{s}$ where Q is the discharge rate and s is the associated drawdown after a long period of time.	If in a fractured aquifer pumping test data are used to determine the non-integer flow dimension n using the Hurst method (See Chapter 2), this n is then substituted into GRF-model and, using curve fitting, a K and S value for the fracture aquifer can be determined (refer to Chapter 2). If in porous aquifer n $\approx$ 2 and the GRF-model will become the Theis equation.
Recharge: Annual rainfall for the rapid assessment and intermediate assessment. Chloride values in rainfall and groundwater for the intermediate and comprehensive assessment. Time series groundwater levels and rainfall data are needed for the comprehensive assessment.	Recharge is calculated directly from Vegter's recharge map (Vegter, 1995).	Recharge is calculated directly from Vegter's recharge map (Vegter, 1995). OR Recharge is calculated using the chloride method (Appendix B, Section B4).	Recharge is calculated using the chloride method (Appendix B, Section B4). OR Recharge is calculated using the EARTH method (Appendix B, Section B4).

Table 4.2 continued

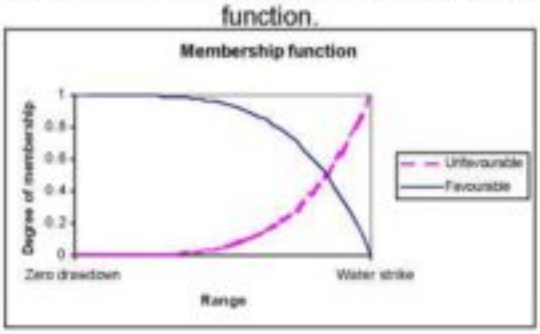
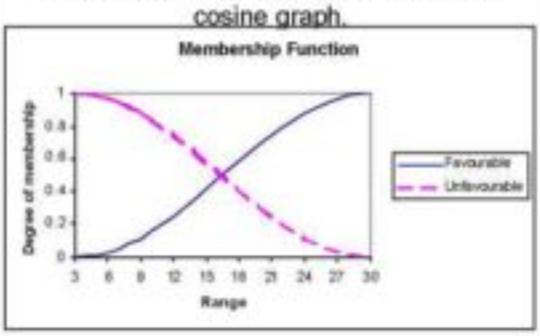
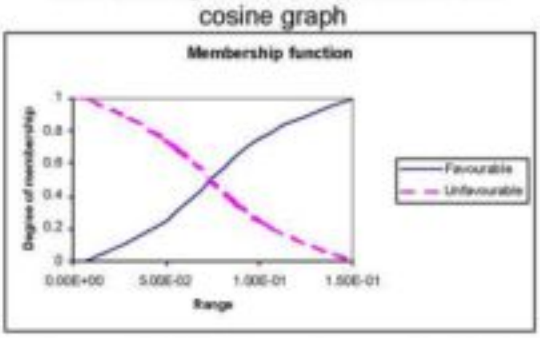
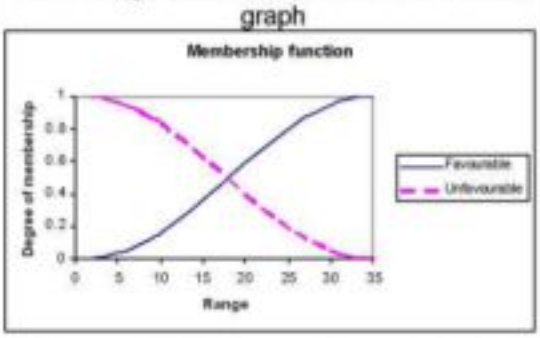
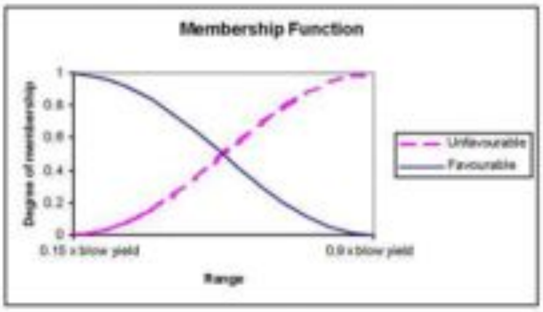
Period (t) for which pumping is going to take place, abstraction rate in borehole under investigation (Q), positions of other abstraction boreholes and abstraction rates, aquifer type and boundary conditions: This information is used to determine the drawdown in the borehole under consideration taking into account other abstraction boreholes and	The drawdown in the borehole under consideration is determined using the Cooper-Jacob method (Kruseman and De Ridder, 1991) $s = \frac{2.3Q}{4\pi T}\log\frac{2.25Tt}{r^2S}$	The drawdown in the borehole under consideration is determined using the Cooper-Jacob method (Kruseman and De Ridder, 1991) $s = \frac{2.3Q}{4\pi T}\log\frac{2.25Tt}{r^2S}$ where r = radius of the borehole and T is determined from slug tests or pumping tests. S is determined from the aquifer type or directly from the user.	The drawdown in the borehole under consideration is determined using the GRF-model (see Chapter 2) $s = \frac{Q}{4\pi^{1-N}K_f b^{3-n} N}\left[\left(\frac{4K_f t}{S_{sf}}\right)^N - \Gamma(1-N)r^{2N}\right]$ Where K_f, S_{sf}, b and r are determined by means of curve fitting. N = 1-n/2, n is determined by means of the Hurst method

other abstraction boreholes and aquifer boundaries. For the rapid assessment only the distances to the closest no-flow boundary and abstraction borehole need to be entered.	where r = radius of the borehole and **T** is determined from rule-of-thumb. S is determined from the aquifer type or directly from the user. The drawdown resulting from the closest abstracting borehole is determined using the same equation except Q is the abstraction rate in the other abstracting borehole, and r is the distance between the closest abstracting borehole and the borehole under investigation. The drawdown as a result of the closest boundary is calculated as in Appendix B, Section B3, Single barrier boundary. The total drawdown in the borehole under investigation is then $s_{total} = s_b + s_c + s$ where s is the drawdown in the borehole not taking into	The drawdown in abstraction borehole i is determined as $s_i = \frac{2.3Q_i}{4\pi T}\log\frac{2.25Tt}{r_i^2 S}$ where Q_i is the abstraction rate in borehole i and r_i is the distance between borehole i and the borehole under investigation. The drawdown in the borehole under investigation which results from the aquifer boundaries is calculated as in Appendix B, Section B3. The total drawdown in the borehole under investigation is then $s_{total} = s_b + \sum s_i + s$ where s is the drawdown in the borehole not taking into account other factors, s_b = drawdown in borehole as a result of the boundaries and $\sum s_i$ = the drawdown in the borehole under investigation as a result of other abstraction boreholes in the area. **Table 4.2** continued	means of the Hurst method. The drawdown in abstraction borehole i is determined as $s_i = \frac{Q_i}{4\pi^{1-N_i} K_{f_i} b_i^{3-n_i} N_i} x \left[\left(\frac{4K_{f_i}t}{S_{sf_i}}\right)^{N_i} \Gamma(1-N_i) r_i^{2N_i}\right]$ Where Q_i is the abstraction rate in borehole i, K_{fi}, S_{sfi}, b_i and r_i are determined by means of curve fitting of pumping test data from borehole i. N_i = 1-n_i/2 and n_i is determined by using the Hurst method, calculated from pumping test data from borehole i. The total drawdown in the borehole under investigation is then $s_{total} = s_b + \sum s_i + s$ where s is the drawdown in the borehole not taking into account other factors, s_b = drawdown in borehole as a result of the boundaries and $\sum s_i$ = the drawdown in the borehole under investigation as a result of other abstraction boreholes in the area.

	account other factors, s_b = drawdown in borehole as a result of the boundary and s_c is the drawdown in the borehole under investigation as a result of the closest abstraction borehole.		
Pumping rate: The user enters the desired pumping rate of the borehole under investigation and the risk of failure is determined for the entered value.			

The membership functions together with their upper and lower limits have been set by experts (G van Tonder from the Institute for Groundwater Studies and R Murray from the CSIR) in the groundwater field and are listed in Table 4.3.

Table 4.3: Membership functions

<table>
<tr><th colspan="4">Membership functions[1]</th></tr>
<tr><td colspan="2">Drawdown
The drawdown membership function is a power function.

</td>
<td colspan="2">Blow yield
The blow yield membership function is a cosine graph.

</td></tr>
<tr><td>Unfavourable limit</td><td>Favourable limit</td><td>Unfavourable limit</td><td>Favourable limit</td></tr>
<tr><td>Water strike/main fracture</td><td>No drawdown</td><td>≤ 3 L/s</td><td>≥ 30 L/s</td></tr>
<tr><td colspan="2">Aquifer type (storativity)
The storativity membership function is a cosine graph

</td>
<td colspan="2">Recharge
The recharge membership function is a cosine graph

</td></tr>
<tr><td>Unfavourable limit</td><td>Favourable limit</td><td>Unfavourable limit</td><td>Favourable limit</td></tr>
<tr><td>1×10^{-5}</td><td>0.15</td><td>≤ 1 %</td><td>≥ 35 %</td></tr>
<tr><td colspan="2">Pumping rate
The pumping rate membership function is a cosine graph.</td>
<td colspan="2">
</td></tr>
<tr><td colspan="2">Unfavourable limit
0.9 x blow yield</td><td colspan="2">Favourable limit
0.15 x blow yield</td></tr>
</table>

[1]The equations for the membership functions are documented in Appendix B, Section B5

The decision rules for all assessments and associated conclusions have been set by G van

Tonder. These rules are listed in Appendix F, Section F1. It is important to note that the decision rules and corresponding conclusions have been set up so that the drawdown carries the most weight.

4.2.5 Example

4.2.5.1 General information

As mentioned in Chapter 2, the Campus test site is located on the grounds of the University of the Free State, South Africa, and covers an area of approximately 180x192 m^2. The aquifer is intersected by thirty percussion and seven core-boreholes (Figure 4-2).

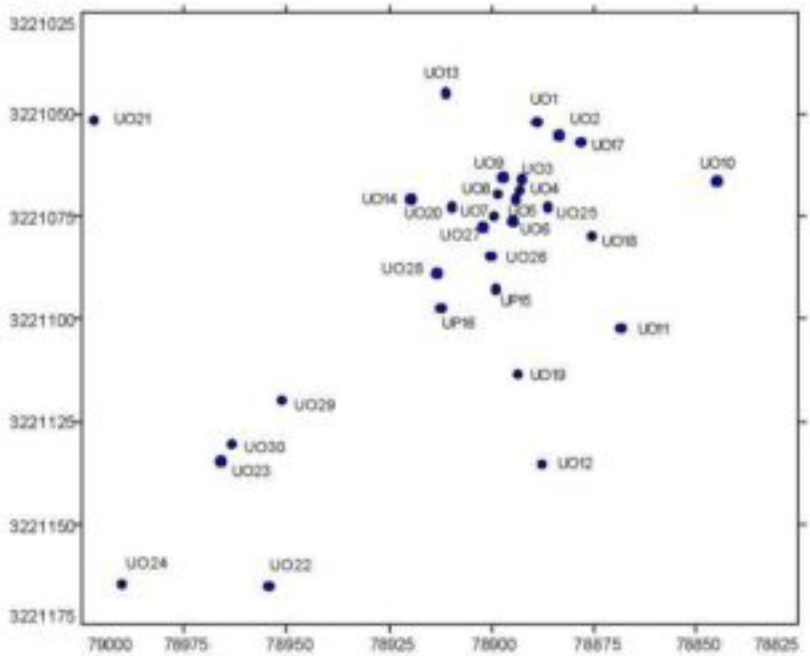


Figure 4-2. The Campus Test Site

The thickness of the aquifer on site is approximately 50m. The aquifer is situated in the Karoo Sequence and the geology consists of sandstone, mudstone and shale deposited under fluvial conditions. Core samples indicate parallel horizontal fractures, the most significant of which is at a depth of 21 m. In more weathered sections of the aquifer, diagonal fractures intersect the bedding plane fractures. The sandstone containing the most horizontal fractures also forms the main water carrying formation. In this analysis the risk of borehole UO5 failing will be determined.

4.2.5.2 Rapid risk assessment

The input values for the rapid assessment are summarised in Table 4.4 and are shown in Figure 4-3. The risk is then determined by selecting "calculate risk".

Table 4.4: Input data for rapid risk assessment

Input	Value
Borehole name	UO5
Pumping rate (L/s)	0.5
Distance to nearest boundary	877.5
Aquifer type	Karoo fractured rock
Water strike (m)	10
Annual rainfall (mm/year)	550
Recharge (mm/year)	Determined by clicking on Vegter's Map
Assessment period (years)	1

Figure 4-3. Rapid assessment screen

The user must note that if the distance to the closest boundary has not been calculated, then the boundary and borehole coordinates can be imported by selecting "geographic data" and the DT will calculate the distance for the user. The risk was calculated for various pumping rates over a period of 1 and 2 years respectively. The results are shown in Figure 4-4. The risk is stated as there is a certain percentage risk that the borehole will dry up over the time span for which the risk is calculated. The higher the risk the higher the chance of the borehole failing.

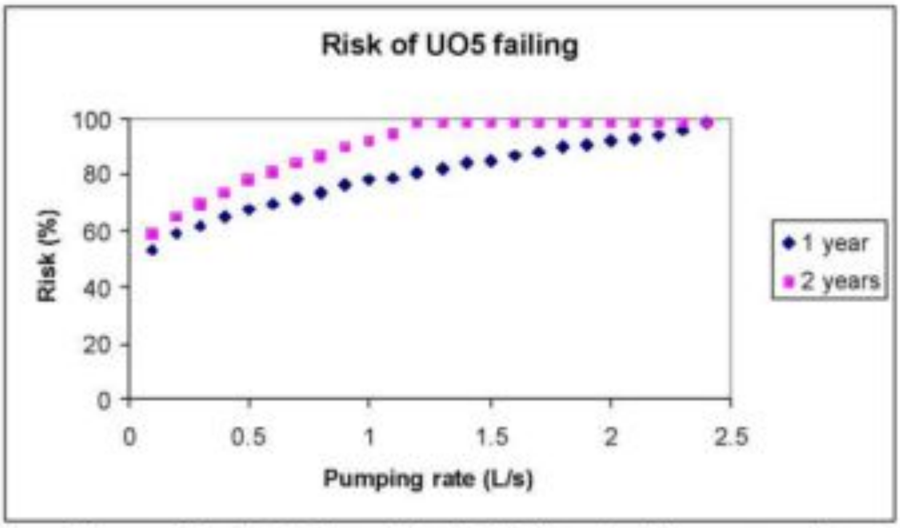


Figure 4-4. Risk of borehole UO5 failing (rapid assessment)

From extensive field investigations it has been proved that UO5 can be pumped for 6 months at 0.33 L/s without failing.

4.2.5.3 Intermediate risk assessment

The data required for the intermediate assessment is summarised in Table 4.5.

Table 4.5: Input data for the intermediate risk assessment

Input	Value
Borehole name	UO5
Borehole coordinates	X = 78893.61 Y = 21071.02
Pumping rate (L/s)	0.5
Boundary type - a - b	Perpendicular 877.5 1755
Aquifer type	Karoo fractured rock
Water strike (m)	10
Annual rainfall (mm/year) Chloride in rainfall (mg/L) Chloride in groundwater (mg/L)	550 1.1 39
Assessment period (years)	1
Pumping test data (to be analysed using the Cooper-Jacob method)	See Appendix B, Section B6 for imported pumping test data

The input screens for the intermediate assessment are shown in Figure 4-5.

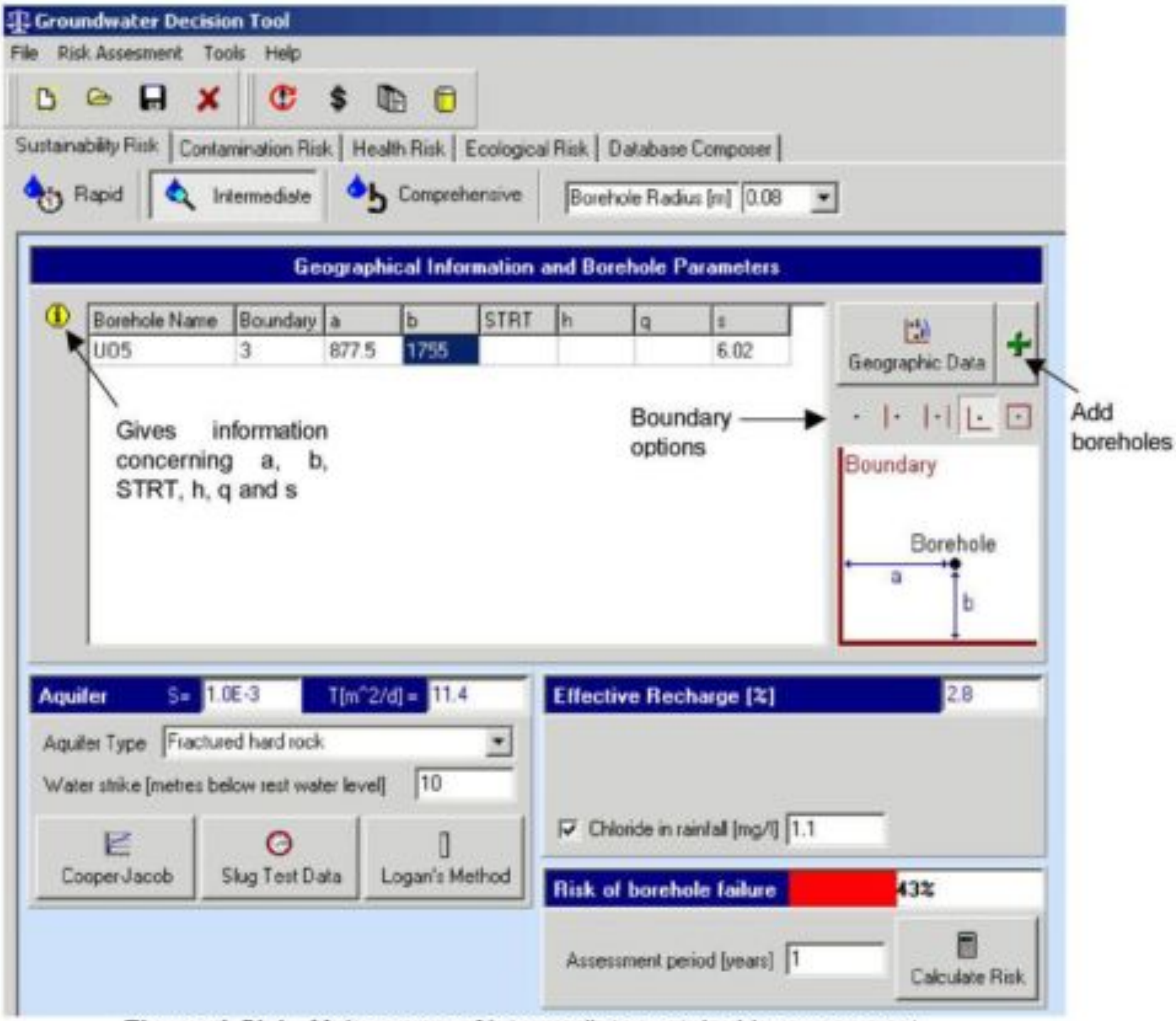


Figure 4-5(a). Main screen of intermediate sustainable assessment

When selecting "Cooper-Jacob", the following screen will appear (Figure 4-5(b)).

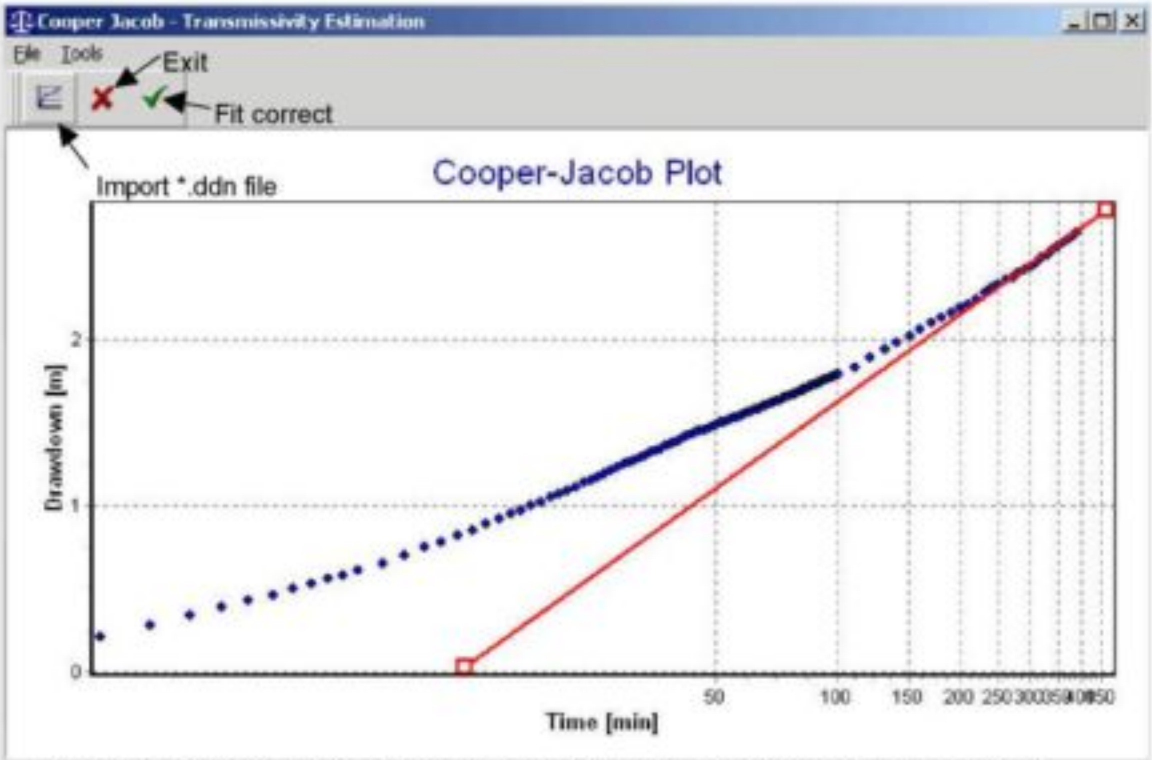


Figure 4-5(b). Screen to fit pumping test data using the Cooper-Jacob method

When selecting "Geographic Data", the following screen appears (Figure 4-5(c)).

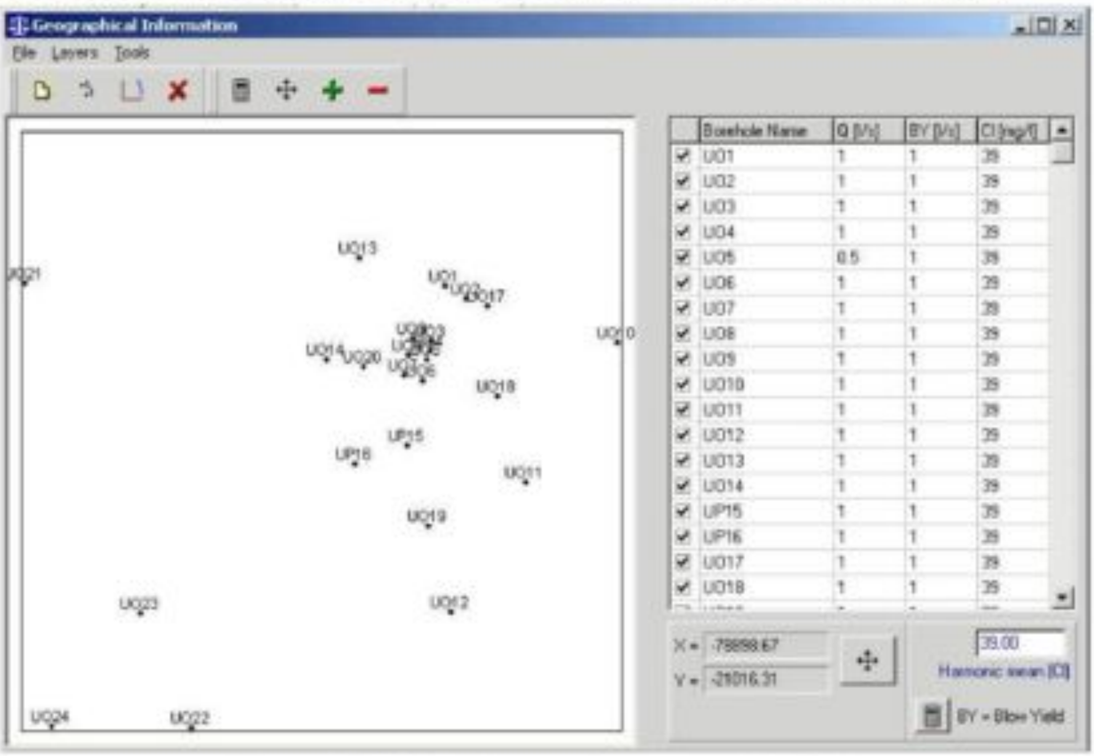


Figure 4-5(c). Input screen for geographical data

All the boreholes have been checked in Figure 4-5(c) to give the user an idea of software capabilities, and that more than one borehole can be assessed at a time. However, in this example only borehole UO5 is under consideration.

Every icon on the toolbar inclues hints to help the user. In Figure 4-5(c) the icons on the toolbar from left to right are: "new", "import boreholes", "import boundaries", "exit", "calculate blow yield", "fit to screen", "add borehole" and "delete borehole". The icons are repeated under the "file" and "tools" menu's. Under the "layers" menu the user can select or deselect the following: "text", "boundaries "and "radii of influence".

The format of the geographical data files (boreholes and boundaries) is documented in Appendix B, Section B7.

The risk of UO5 failing was calculated over a period of 1 and 2 years for various pumping rates (Figure 4-6).

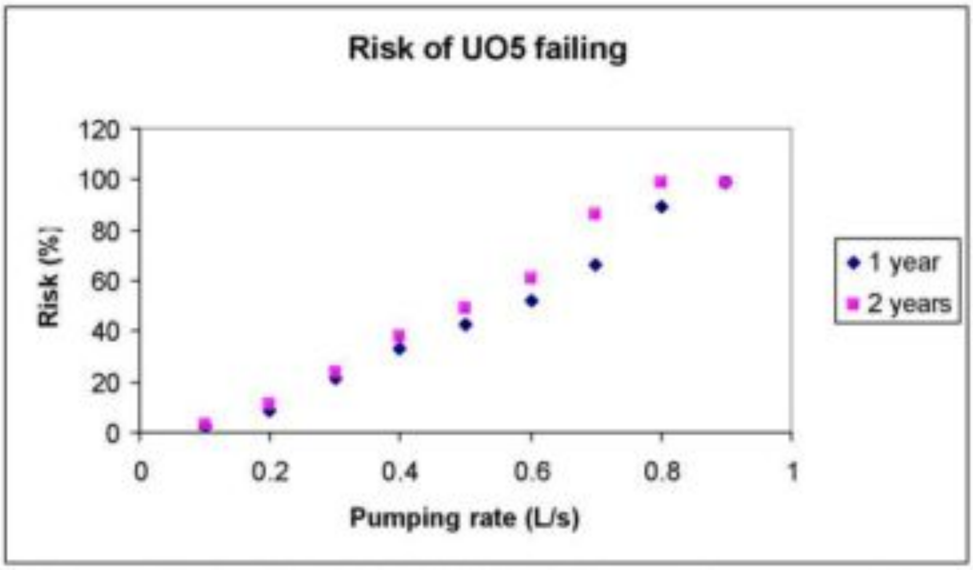


Figure 4-6. Risk of borehole UO5 failing (intermediate assessment)

4.2.5.4 Comprehensive risk assessment

The data required for the comprehensive assessment is summarised in Table 4.6.

Table 4.6: Input data for the comprehensive risk assessment

Input	Value
Borehole name	UO5
Borehole coordinates	Imported from *.bhl file. See Appendix B, Section B7
Pumping rate (L/s)	0.5
Boundary type - a - b	Perpendicular 877.5 1755
Aquifer type	Karoo fractured rock
Water strike (m)	10
Annual rainfall (mm/year) Chloride in rainfall (mg/L) Chloride in groundwater (mg/L)	550 1.1 39
Assessment period (years)	1
Pumping test data (to be analysed using the Hurst method and GRF-model)	See Appendix B, Section B6 for imported pumping test data

The input screens for the assessment are shown in Figure 4-7.

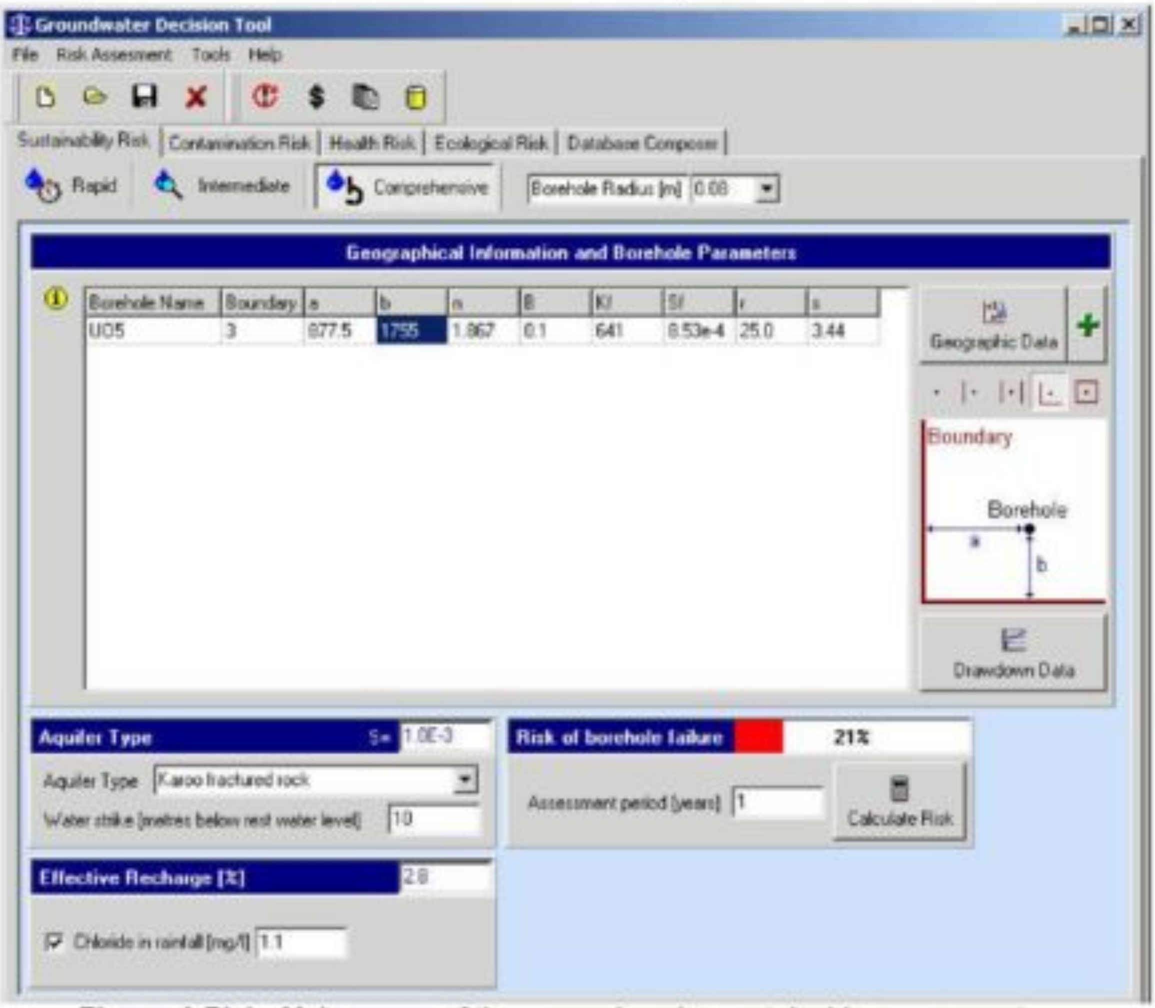


Figure 4-7(a). Main screen of the comprehensive sustainable assessment

When selecting "drawdown data" the following screen (Figure 4-7(b)) appears:

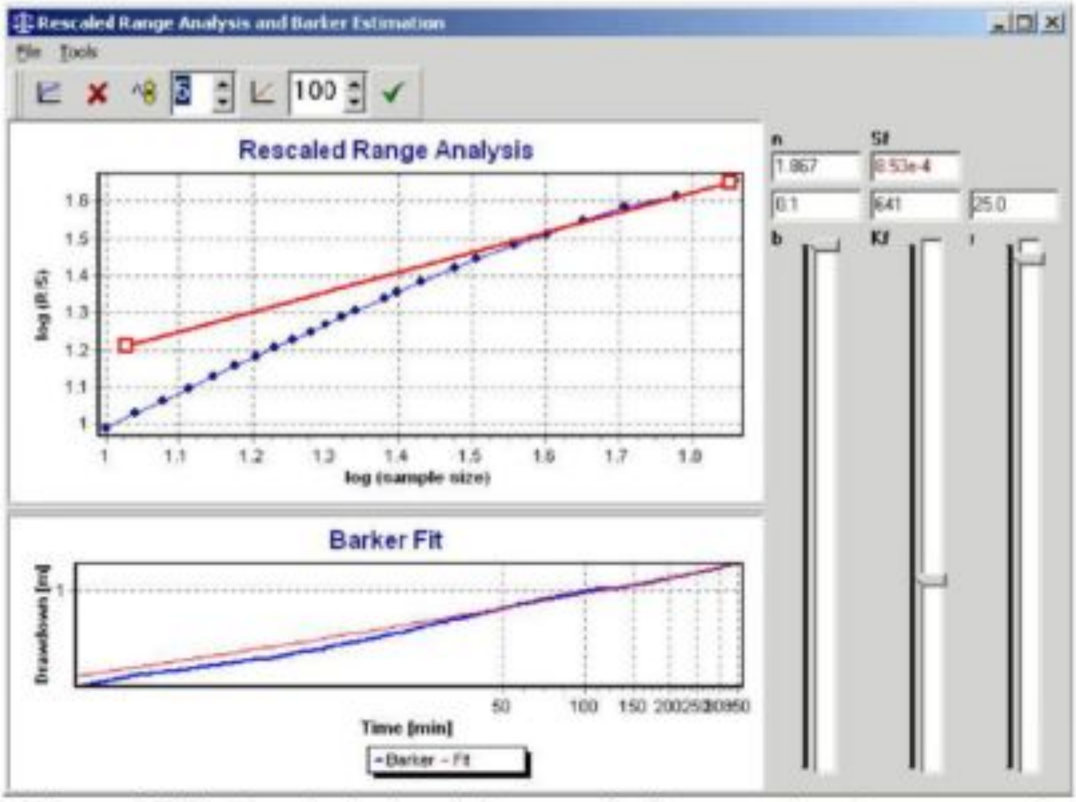


Figure 4-7(b). Pumping test analysis screen for the comprehensive assessment

The results of the assessment for various pumping rates over a period of 1 and 2 years respectively are shown in Figure 4-8.

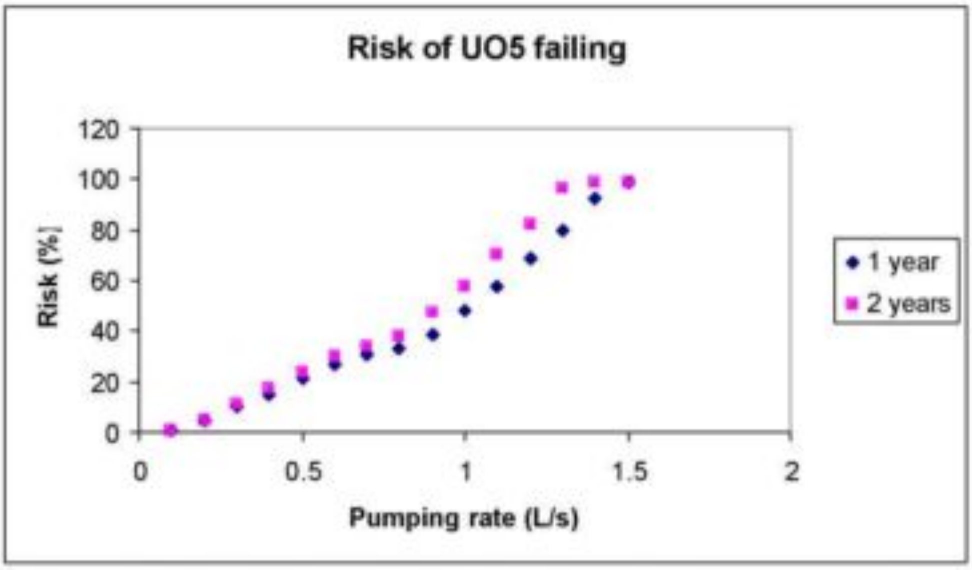


Figure 4-8. Risk of borehole UO5 failing (comprehensive assessment)

Figure 4-9 shows the results of the rapid, intermediate and comprehensive assessment after 2 years. The rapid assessment does not take into account all the boundaries and uses elementary methods to calculate values, therefore the risks are lower than those of the intermediate assessment. The intermediate assessment basically uses the transmissivity of the matrix to calculate the risks, and its risks are therefore higher than the comprehensive assessment which takes the fracture into account.

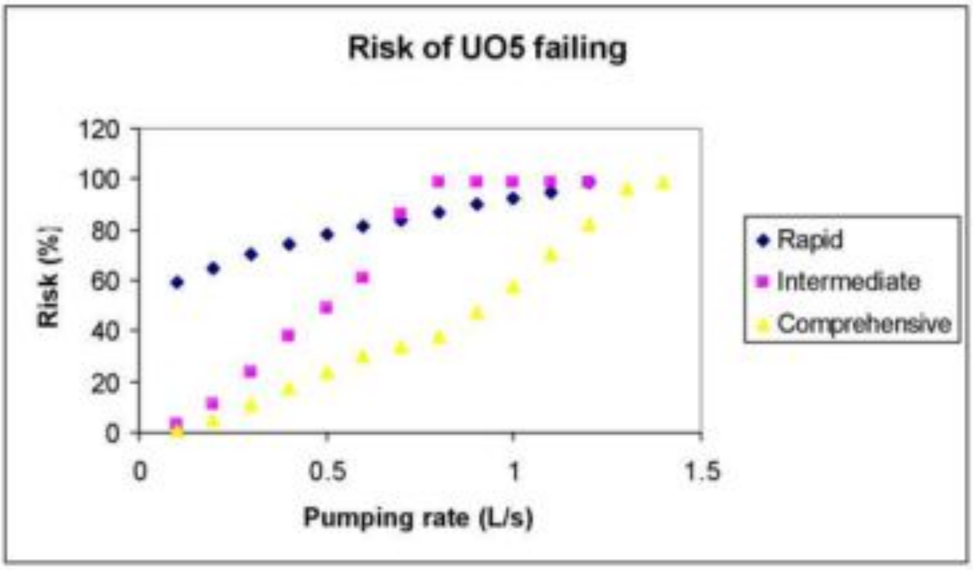


Figure 4-9. Comparison of risks after pumping for 2 years for the rapid, intermediate and comprehensive assessments.

NOTE:

- The Earth model has not been used in this assessment, although it is also a curve fitting method. The format for the required data file is documented in Appendix B, Section B8.
- The radius of influence of the borehole under investigation can be viewed after the risk has been calculated, the user just has to select "geographic data".

CHAPTER 5

Groundwater Contamination Risk Assessment

5.1 GENERAL

Contamination releases to groundwater can occur by design, by accident, or by neglect. Most groundwater contamination incidents involve substances released at or slightly below the surface of the earth. The protection of groundwater quality is complex and, as groundwater is affected by virtually every activity of society, it is difficult to develop and implement effective methodologies to determine groundwater contamination risks. In addition, many potential hazardous contaminants are colourless, odourless and tasteless and therefore difficult to detect by passive means (Barcelona *et al.*, 1988). In spite of all these problems, a comprehensive, integrated approach to groundwater protection is essential if groundwater quality standards for the highest beneficial use are to be met and maintained (Lynch *et al.*, 1994).

Not all land-use activities pose the same contamination threat to groundwater resources and different parts of the environment have varying capacities for dealing with contamination. Consequently, it is necessary to review the risk of groundwater contamination in two separate but interrelated ways, namely considering the characteristics of the contaminant and considering the vulnerability of the aquifer to contamination. The characteristics of the contaminant include the source of the contaminant, its loading as well as information about the contaminant itself. Aquifer vulnerability, on the other hand, represents the intrinsic characteristics that determine the sensitivity of an aquifer to the adverse effects resulting fom the imposed contaminant (Lynch *et al.*, 1994). The groundwater contamination risk assessment therefore consists of a vulnerability assessment and a contaminant assessment.

PLEASE NOTE: Contamination and pollution have both been referred to in this chapter and in this context they both refer to the direct or indirect alteration of the chemical properties of a water resource as to make it harmful or potentially harmful to human beings and/or aquatic ecosystems.

5.1.1 Aquifer Vulnerability

An approach similar to that of DRASTIC (Rosen, 1994) was followed to determine the vulnerability of an aquifer. The parameters needed for describing vulnerability are:

- Depth to groundwater: this gives an indication of the distance and time required for the contaminant to move through the unsaturated zone to the aquifer.
- Recharge: the primary source of groundwater is precipitation which aids the movement of a contaminant to the aquifer.
- Aquifer media: the consolidated or unconsolidated rock matrices that serve as water-bearing units. In this approach, the fractures that occur in the rock matrix will also be taken into account.
- Soil media: this consists of the upper portion of the vadose zone (Aller *et al.*, 1987). The soil media can affect the rate at which contaminants migrate to groundwater.
- Topography: will give an indication on whether a contaminant will run off or remain on the surface long enough to infiltrate into the groundwater.
- Impact of the vadose zone: this is defined as that portion of the geological profile beneath the earth's surface and above the first principal water-bearing aquifer (Lynch *et al.*, 1994). The vadose zone can retard the progress of the contaminant.

5.1.2 Contaminant Assessment

The following information regarding the contaminant and the effects thereof are taken into account in the contamination assessment:

- Contaminant: As the National Water Act (Act No 36, 1998) emphasises the importance of basic human needs and aquatic ecosystems, the drinking water guidelines (DWAF, 2001) and aquatic ecosystem guidelines (South African Water Quality Guidelines, 1996) will be used as a basis to determine the potential impacts of a certain contaminant.
- Duration of contamination: if the contamination results from a single (once-off) spill, the impact will probably be smaller than that resulting from continuous contamination.
- Contaminant properties include aspects such as:-
    - Matrix diffusion: the process of solute diffusion from fractures with high solute concentrations to the rock matrix, which has a lower solute concentration.
    - Dispersion: a measure of the spreading of a flowing substance due to the nature of the rock matrix with its interconnected channels distributed randomly in all directions.

Figure 5-1 summarises the steps in the groundwater contamination risk assessment.

5.2 DATA REQUIREMENTS

According to the definition of a rapid assessment, low levels of confidence are attached to the results and no intensive field investigations are necessary. Therefore the data requirements are limited and the assessment will rely heavily on data from the datasheets stored in the database of the decision tool. At least one field investigation is attached to the intermediate assessment. For high confidence results, a comprehensive assessment has to be completed, for which good quality data, based on intensive field investigations, is necessary. Table 5.1 contains the data required for a contamination risk assessment.

Groundwater contamination risk assessment

Problem formulation

Hazard identification establishes whether exposure to a contaminant can cause groundwater contamination. Once a hazard has been identified, the remainder of the process encompasses the description of the properties of the hazardous agent and the vulnerability of the aquifer.

Questions to be asked: What is the problem? Why is it a problem?

Analysis

Aquifer Vulnerability

Establishes the tendency or likelihood for contaminants to reach a specified position in the groundwater system after introduction at some location above or in the aquifer.

Contaminant Assessment

Establishes the frequency, duration and potential impacts (on human health and aquatic ecosystems) of contamination, taking into account the properties of the contaminant

Risk characterisation

Provides an indication of the incidence of possible contamination taking into account properties of both the aquifer and the contaminant. Risk characterisation should include information that is useful to both stakeholders and risk managers.

Figure 5-1. Summary of methodology for contamination risk assessment

Table 5.1: Data required for contamination risk assessment and potential data sources

Data required	**Potential data sources**		
	Rapid	**Intermediate**	**Comprehensive**
Vulnerability Assessment			
• Depth to groundwater	User	User	User
• Recharge	Map[1]/User	Map[1] OR	
○ Rainfall	-	User	User
○ Chloride in rainfall	-	Laboratory analyses	Laboratory analyses
○ Chloride in groundwater	-	Laboratory analyses	Laboratory analyses
○ Water levels in boreholes	-	-	OR Field data
• Aquifer media	Geological map/User	Borehole logs/augering	Borehole logs
• Soil media	Soil map[2]/User	Soil map[2]/Laboratory analyses	Laboratory analyses
• Topography	Topographical map/User	Topographical map/Field data	Field data
• Vadose zone information	DT database[3]	DT database[3]/Field data	Field data
Contaminant Assessment			
• Contaminant	User	User	User
• Position of source	-	User	User
• Concentration at source	-	Laboratory analyses	Laboratory analyses
• Contaminant injection rate	-		User
• Drinking water guidelines	DT database[4]	DT database[4]	DT database[4]
• Aquatic ecosystem guidelines	-	DT database	DT database
• Position of boreholes	-	User	User
• Abstraction rates	-	User	User
• Diffusion/matrix diffusion values	DT database[5]	DT database[5]/experiments	Experiments
• Dispersivity values			Calibration of equations
○ Migration distance	DT database[6]/User	DT database[6]/User	
• Groundwater gradient	-	User	User
• Hydraulic conductivity	-	DT database[8]/user	User to determine from pumping tests
• Thickness of aquifer	-	-	User
• Area over which contamination is taking place	-	-	User
• Porosity	-	DT database[9]/user	User from tracer test or laboratory experiments
• Duration of contamination	User to choose from DT database[7]	User	User

-Data not needed for assessment. [1]Map included in DT (see Appendix B, Section B1).
[2]Map included in DT (see Appendix C, Section C1). [3]See Table 5.3.
[4]See Appendix C, Section C2. [5]See Appendix C, Section C3.
[6]See Appendix C, Section C4. [7]See Table 5.3.
[8]See Appendix C, Section C7. [9]See Appendix C, Section C8.

5.3 ASSUMPTIONS AND LIMITATIONS

Rapid Assessment

For the rapid assessment, no chemical analyses are needed and the concentrations of the contaminants are therefore not included in the assessment. If there are also no field observations for the data discussed in Table 5.1, then values from maps or the literature recorded in the DT database can be used. However, such an assessment cannot be used to make accurate conclusions concerning the contamination risk.

Only drinking water guidelines are used in this assessment.

Intermediate Assessment

This is a medium confidence assessment based on limited field data. In general, no time series data will be available. When calculating the concentration of contamination at any point in the aquifer, only advective-dispersive processes are taken into account.

The aquatic ecology and drinking water guidelines are considered in the intermediate assessment.

Comprehensive Assessment

A comprehensive assessment is a quantitative assessment and high confidence is attached to the results. Intensive field investigations are therefore necessary. Even with all the field data, it is impossible to record and include all heterogeneities present in the aquifer.

Laboratory experiments and tracer tests must be conducted to determine accurate contaminant parameters such as matrix diffusion and porosity. Tracer tests can also be used to determine the influence of fractures in the movement of contamination. Two types of tracer tests are discussed in Chapter 2. The analyses of these are not included in the DT tool, however the Institute for Groundwater Studies is developing software to analyse these tests, and this will shortly be available.

The aquatic ecology and drinking water guidelines are taken into account in the comprehensive assessment. The equations that are used to calculate contaminant concentrations assume that there is a uniform flow rate through the aquifer.

General – applicable to all assessments

Due to the shortage of information concerning membership functions, the functions will be presumed to be cosine or specific values will be assigned to specific conditions. The methodology only takes into account chemical contamination, specifically matrix diffusion and dispersion. All other chemical processes/reactions are ignored.

If there are no impacts on the ecology or human health, the risk of contamination is assumed to be zero.

5.4 METHODOLOGY

The methodology is the same as discussed in Section 3.2. Therefore it is once again necessary to set up decision rules and membership functions, which are the same for all tiers of the assessment. However the quality of data and calculation of concentrations vary for the three different assessments. The recharge and concentrations calculations needed for the assessments are discussed in Table 5.2.

The vulnerability and contaminant assessment membership functions/values are listed in Tables 5.3 and 5.4 respectively. Where possible the DRASTIC ratings have been used (Lynch *et al.*, 1994), but where necessary experts (B Usher and G van Tonder from the Institute for Groundwater Studies, University of the Free State) in groundwater contamination were consulted to establish the decision rules and membership functions/values.

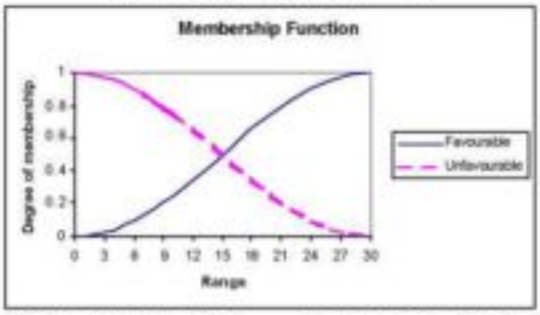
Table 5.2: Calculations needed for the contamination assessment

Data required for calculations	Calculations		
	Rapid assessment	Intermediate assessment	Comprehensive assessment
Recharge: Annual rainfall for both the rapid and intermediate assessments. Chloride values in rainfall and groundwater for the intermediate and comprehensive assessments. Time series groundwater levels and rainfall data for the comprehensive assessment.	Recharge is calculated directly from Vegter's recharge map (Vegter, 1995).	Recharge is calculated directly from Vegter's recharge map (Vegter, 1995). OR Recharge is calculated using the chloride method (Appendix C, Section C4).	Recharge is calculated using the chloride method (Appendix C, Section C4). OR Recharge is calculated using the EARTH method (Appendix C, Section C4).
Concentration: Determines the impacts of the contaminant on humans and the ecology. No information is needed for the rapid assessment. For the intermediate assessment the following is needed: • Initial concentration at the source (C_0). • Length between borehole and source (L) or source coordinates. • Migration distance to calculate dispersivity values (α). • Duration of contamination (t). • Groundwater gradient (dh/dl), effective porosity (n_e) and hydraulic conductivity (K) to determine velocity (v). • Dispersivity (α) and velocity (v) to determine the dispersion coefficient (D). For the comprehensive assessment the following is needed: • Initial concentration at the source. • Positions of source and boreholes. • Duration of and injection rate of contaminant. • Area of source and thickness of aquifer. • Groundwater gradient, effective porosity and hydraulic conductivity to determine velocity. • Dispersion coefficients. • Concentration at point in aquifer together with coordinates of point and time at which the concentration was measured.	-	The equation used to calculate concentrations is in the following format (Fetter, 1999): $C = \frac{C_0}{2}\left[\operatorname{erfc}\left(\frac{L - vt}{2\sqrt{Dt}}\right)\right]$ where $D = \alpha v$ and $v = \frac{K}{n_e}\frac{dh}{dl}$.	Two calculations will be used for the following contamination sources: • Continuous injection • Slug (once-off) injection For information concerning the equations used in this section refer to Appendix C (Section C5). It is important to note that dispersion coefficients must be determined by means of the equations in Appendix C (Section C5).

Table 5.3: Membership functions/values for decision rules for vulnerability assessment

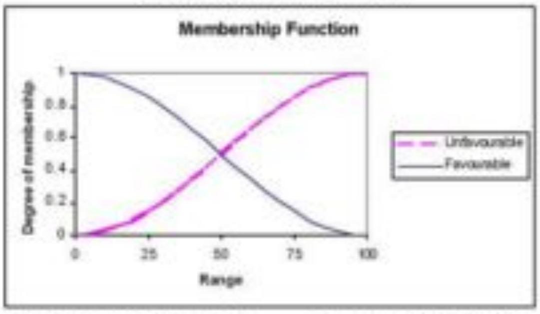
Vulnerability membership functions/values

Depth to groundwater[1] (m)



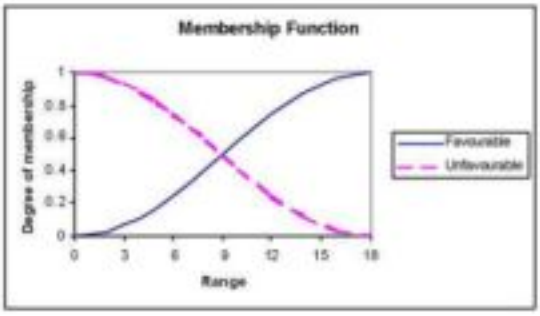
Unfavourable limit	Favourable limit
0	30

Recharge[1] (mm/year)



Unfavourable limit	Favourable limit
100	0

Topography[1] (%)



Unfavourable limit	Favourable limit
0	18

Soil media[1]

Type[2]	Membership
SaCl, SaCl-Cl	0.6
SaClLm-Cl, SaClLm-SaCl	0.6
SaClLm, SaLm-SaCl	0.5
SaLm-SaClLm	0.5
SaLm	0.4
Sa-LmSa, SaLmSa etc	0.35
Sa-SaLm, LmSa-SaLm, LmSa	0.3
Sa	0.0

Aquifer Media[1]

Type	Membership
Dolomite (massive)	1.0
Intergranular	0.2
Fractured	0.4
Fractured and weathered	0.7
Dolomite (karstic)	0.0

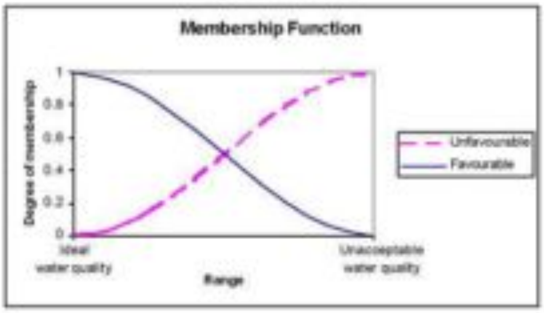
Impact of the vadose zone[1]

Type	Membership
Beach sands and Kalahari	0.0
Dolomite	0.1
Table Mountain, Witteberg, Granite, Natal, Witwatersrand, Rooiberg, Greenstone, Dominion, Jozini	0.4
Karoo (southern)	0.5
Ventersdorp, Pretoria, Griqualand West, Malmesbury, Van Rhynsdorp, Uitenhage, Bokkeveld, Basalt, Waterberg, Soutpansberg, Karoo (northern), Bushveld, Olifantshoek	0.6
Gneiss, Namaqua metamorphic rocks	0.7

[1]Taken from Lynch *et al.* (1994). [2]Sa = sand, Lm = loam, Cl = clay.

The decision rules, taking into account the above parameters for the vulnerability assessment, are documented in Appendix F (Section F2). The aquifer vulnerability risk is then calculated using Equation 3.1.

Table 5.4: Membership functions/values for decision rules for contaminant assessment

<table>
<tr><th colspan="3">Contaminant assessment membership functions/values</th></tr>
<tr><th>Contaminant[1]</th><th colspan="2">Contamination duration</th></tr>
<tr><td rowspan="7">

Unfavourable limit Unacceptable water
Favourable limit Ideal water quality</td><td>Type</td><td>Membership</td></tr>
<tr><td>Contamination may be seconds, minutes or hours</td><td>0.9</td></tr>
<tr><td>Contamination occurs at intermittent periods < 2 years</td><td>0.6</td></tr>
<tr><td>Contamination > 90 days and < 2 years</td><td>0.6</td></tr>
<tr><td>Contamination occurs at intermittent periods >2 years</td><td>0.3</td></tr>
<tr><td>Continuous contamination > 2 years</td><td>0.0</td></tr>
<tr><td></td><td></td></tr>
<tr><th colspan="3">Contaminant properties</th></tr>
<tr><th>Range[2]</th><th colspan="2">Membership value</th></tr>
<tr><td>Very high</td><td colspan="2">0.2</td></tr>
<tr><td>High</td><td colspan="2">0.4</td></tr>
<tr><td>Medium</td><td colspan="2">0.6</td></tr>
<tr><td>Low</td><td colspan="2">0.8</td></tr>
<tr><td>Very low</td><td colspan="2">1.0</td></tr>
</table>

[1]Taken from Quality of Domestic Water Supplies (2001). Appendix C (Section C2) contains more information. For the rapid assessment, concentrations are not needed therefore the contaminants are classified according to death (0.0), effects common (0.25), long-term effects (0.4) and few effects (0.8).

[2]For the calculation of the ranges refer to Appendix C (Section C6).

The risk of contamination based on the contaminant characteristics is calculated using Equation 3.1. The decision rules on which the calculations are based are listed in Appendix F (Section F2).

The overall groundwater contamination risk assessment can then be calculated by means of the following condition:

$$\text{Risk}_{\text{contamination}} = \max(\text{Risk}_{\text{vulnerability}}, \text{Risk}_{\text{contaminant}})$$

5.5 EXAMPLE

The Campus Test Site is once again taken as an example for the contamination assessment. A vulnerability assessment will be conducted to determine the risk of contamination based on the aquifer properties. All three tiers of the assessment are similar except for the method by which the recharge is calculated and the quality of data required.

As the recharge calculations were discussed in the sustainable assessment, only the rapid vulnerability risk assessment will be demonstrated here.

The input for the vulnerability risk is summarised in Table 5.5.

Table 5.5: Input data for rapid vulnerability risk assessment

Input	Value
Recharge (mm/year)	Select from map
Soil media	Select from map
Aquifer media	Fractured
Vadose zone	Karoo (southern)
Groundwater depth (mbgl)	10
Topography (% slope)	0.3

The results of the risk assessment are shown in Figure 5-2.

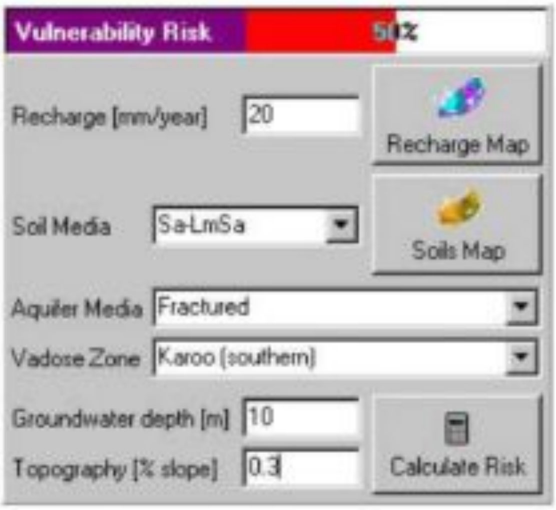


Figure 5-2. Results of rapid vulnerability assessment

It is important to note that this risk in the rapid assessment might differ slightly from that in the intermediate and comprehensive assessments as the quality of data required for the latter two assessments is better than that required for the rapid assessment.

There is a difference between the rapid assessment, and the intermediate and comprehensive contaminant assessments. Therefore, an example of both the rapid and intermediate assessments will be discussed.

For the rapid contaminant assessment, the example of a spill of tricholoroethane will be considered. Trichloroethane, also known as methyl chloroform, does not occur naturally in the environment. It is found in many common products such as glue, paint, industrial degreasers and aerosol sprays. The maximum drinking water guideline is 0.2 mg/L.

The data needed for the rapid assessment are listed in Table 5.6.

Table 5.6: Input data for rapid contaminant risk assessment

Input	Value
Duration of contamination	Polluting > 90 days and < 2 years
Contaminant	tricholoroethane
Matrix diffusion value[1] (m^2/s)	1.01×10^{-9}
Dispersivity[2] (m)	50

[1]This value is assumed and has not been calculated. A value can also be chosen from the DT database. [2] Migration distance can also be given and then DT will calculate dispersivity.

The results of the rapid risk assessment are shown in Figure 5-3.

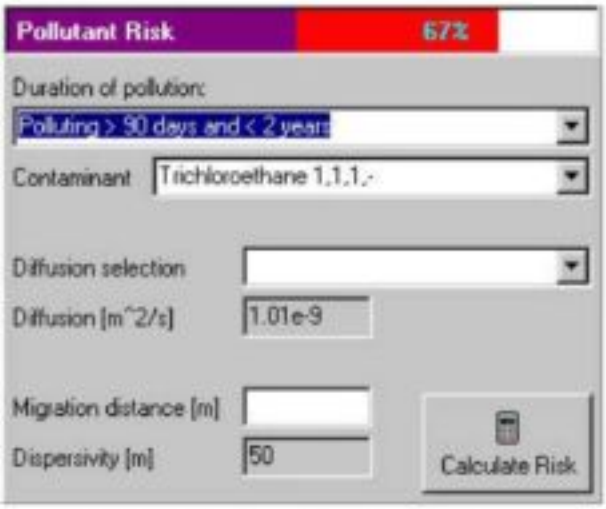


Figure 5-3. Results of rapid contaminant assessment

The total rapid contamination risk is calculated as:

$$Risk_{contamination} = \max(Risk_{vulnerability}, Risk_{contaminant}) = 67\%$$

As the Campus Test Site has a fracture at 21 m, two scenarios will be completed for the intermediate contaminant assessment, the first being the movement of the contaminant through the fracture, and the second the movement of the contaminant through the sandstone matrix. The risk of borehole UO5 being contaminated is calculated assuming borehole UO5 is not being pumped. In addition, there are no aquatic ecosystems in the vicinity, so the risk will be calculated based on drinking water guidelines. For the intermediate contaminant risk assessment, the data documented in Table 5.7 was used. A concentration of 10 mg/L was assigned at the source.

Table 5.7: Input data for the intermediate contaminant risk assessment

Input	Fracture	Matrix
Position of source	X = -78393.61 Y = -21571.02	X = -78393.61 Y = -21571.02
Position of borehole UO5	X = -78893.61 Y = -21071.02	X = -78893.61 Y = -21071.02
Contaminant	tricholoroethane	tricholoroethane
Concentration at source (mg/L)	10	10
Matrix diffusion value[1] (m^2/s)	1.01×10^{-9}	1.01×10^{-9}
Porosity (%)	49	6
Hydraulic conductivity (m/d)	200	2
Groundwater gradient	0.003	0.003
Duration of contamination	Polluting > 90 days and < 2 years	Polluting > 90 days and < 2 years

[1]This value is assumed and has not been calculated. A value can also be chosen from the DT database.

The results of the assessment are listed in Table 5.8. It is clear that the contaminant moves faster along the fracture zone. The DT holds the risk at 12% due to the contamination duration being continuous and the contaminant properties not being favourable. Once the contaminant reaches borehole UO5 the risk start increasing. An example of the intermediate assessment screen is shown in Figure 5-4.

Table 5.8: Results of intermediate contaminant assessment

Time (years)	Risk (%)	
	Fracture	Matrix
0.25	12	12
0.5	23	12
0.625	69	12
0.75	99	12
6	99	21
7	99	50
8	99	98
8.5	99	99

The "plus" sign buttons in Figure 5-4 are used to add contaminants for the assessment. If more than one contaminant is selected, the DT will calculate the risk for each one. The final risk will then be the maximum of the individual risks calculated. The input screen for the geographical data is similar to that used in the sustainable risk assessment except that the source is depicted in a red square.

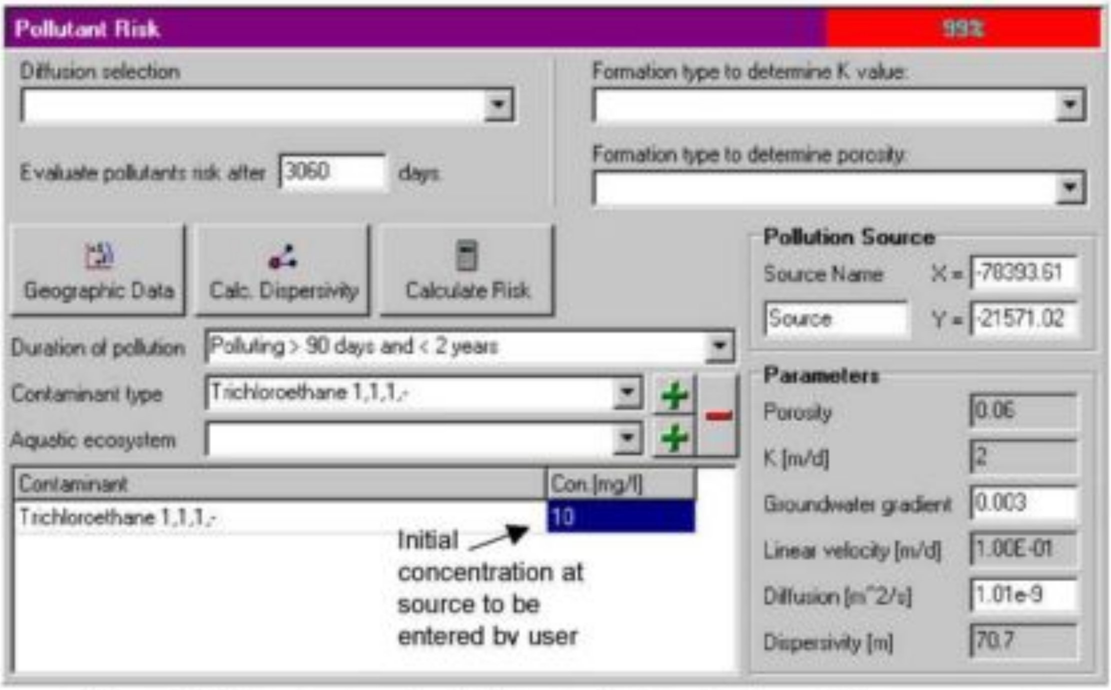


Figure 5-4. Input screen for the intermediate contaminant risk assessment

The comprehensive contaminant assessment is almost identical to the intermediate assessment except, that the user can select either a slug source or a continuous source. It is important to note that with the slug source the contamination moves through the aquifer as a pulse and once the maximum concentration has passed a point in the aquifer, the risk will start decreasing at that point. The dispersivity is calculated by means of a mass transport equation documented in Appendix C, Section C5.

5.6 MINES

5.6.1 Background

There are basically two types of mines found in South Africa, namely open cast mines and underground mines. Underground mining can be assessed with the DT, but open cast mines can overflow. The water quality of these mines can have a major impact on human health and the environment. This poor quality water usually flows down gradient toward surface water bodies. The sulphate values in the water are usually high and can cause dehydration and diarrhoea in humans.

5.6.2 Methodology

Open cast mines, the majority of which are coal mines in South Africa, fill up with water and decant after closure. In the South African coal mines this normally occurs within 10 years of closure. At the more isolated collieries, rebound of groundwater can take up to 50 years (Grobbelaar, 2000).

As the quality of decanting groundwater is important, it will be discussed in this section and included in the decision tool. No fuzzy logic calculations are required as the decant rate is a relatively easy to calculate. The drinking water guidelines specified for sulphates will be used to determine the risks associated with the quality of decanting water.

The data requirements for determining the amount of water that will decant and the quality thereof are listed in Table 5.9.

The amount of water that will decant is calculated with the following Equation (Van Tonder, 2001(a)):

$$\text{Decant rate} = (\text{R} \times \text{area of mine}) + \text{I} - \text{O}$$

where

R = Effective recharge

I = Inflow of groundwater into the mine

O = Outflow of groundwater out of the mine

The time taken for decanting to start once mining operations have ceased can be calculated as:

$$\text{Time} = \frac{\text{volume of open cast mine} \times \text{storavity of spoils}}{\text{I} + \text{R}}$$

The concentration of sulphate present in groundwater decanting can then be determined as:

$$\text{Concentration} = \frac{\text{SO}_4 \text{ generation} \times \text{area of mine}}{\text{decant rate}}$$

The amount of mixing in a river can be determined as:

$$\text{Mixing} = \frac{\text{Load of sulphate at river}}{\text{Low flow in river}}$$

where the load of sulphate at the river is calculated as:

$$\text{Load of sulphate at river} = \text{Concentration} \times \text{decant rate}$$

Table 5.9: Data required for determining decanting and possible data sources

Data required	Rapid assessment	Intermediate assessment	Comprehensive assessment
Storativity of spoils	Assumed to be 25%	Assumed to be 25%	User
• Recharge o Rainfall o Chloride in rainfall o Chloride in gw o Water levels in boreholes	Map[1]/User - - - -	Map[1] OR User Laboratory analyses Laboratory analyses -	 User Laboratory analyses Laboratory analyses OR Field data
Area of mine Depth of mine	User User	User User	User User
Flow into and out of the mine • Groundwater gradient • Hydraulic conductivity • Aquifer thickness • Transmissivity - Pumping test data - Slug test data • Length of mine circumference where groundwater is flowing into the mine • Length of mine circumference where groundwater is flowing out of the mine	 User User/DT database[2] User - - - User User	 User - - User to be determined using Logan's method OR User User User	 User to be determined from Bayesian interpolation[3] - - User to be determined using Cooper-Jacob method - User User
Amount of sulphate generated	Assume to be 7 kg/d/ha (Hodgson, 2001)	Assume to be 7 kg/d/ha (Hodgson, 2001)	Field data
Low flow • Quaternary catchment • Low flow	 User	 User	 User OR User

[1]Refer to Appendix B, Section B1.
[2]See Appendix C, Section C7.
[3]Can be determined using software package Tripol, developed at the Institute for Groundwater Studies. This software can be accessed via the DT.

The risks can then be determined based on drinking water standards. These are determined by means of a cosine function (see Figure 5-5).

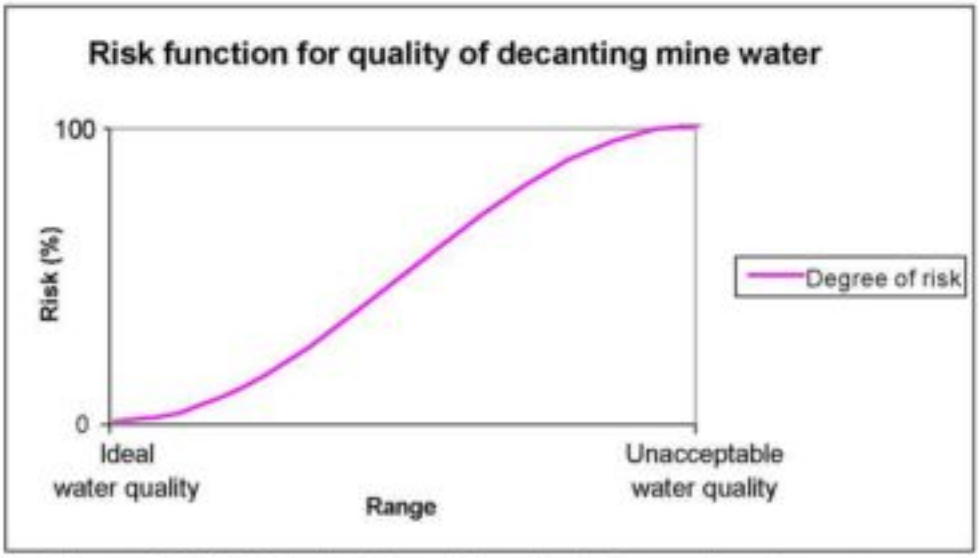


Figure 5-5. Risk function for sulphate load in decanting mine water

5.6.2 Example

The three tiers of assessment are similar and therefore a hypothetical open cast mine with the parameters listed in Table 5.10 will be used for the rapid assessment to calculate the risk of contamination in the river as a result of decanting. The input screen and results of the risk assessment are shown in Figure 5-6.

Table 5.10: Data needed for the open cast mine contamination risk assessment

Input	Value
Length of outflow (m)	500
Length of inflow (m)	500
Depth of mine (m)	25
Area of mine (ha)	25
Groundwater gradient	0.02
Transmissivity (m^2/d)	13
Quaternary catchment	B41B
Recharge determined from Vegter's map (mm/year)	32

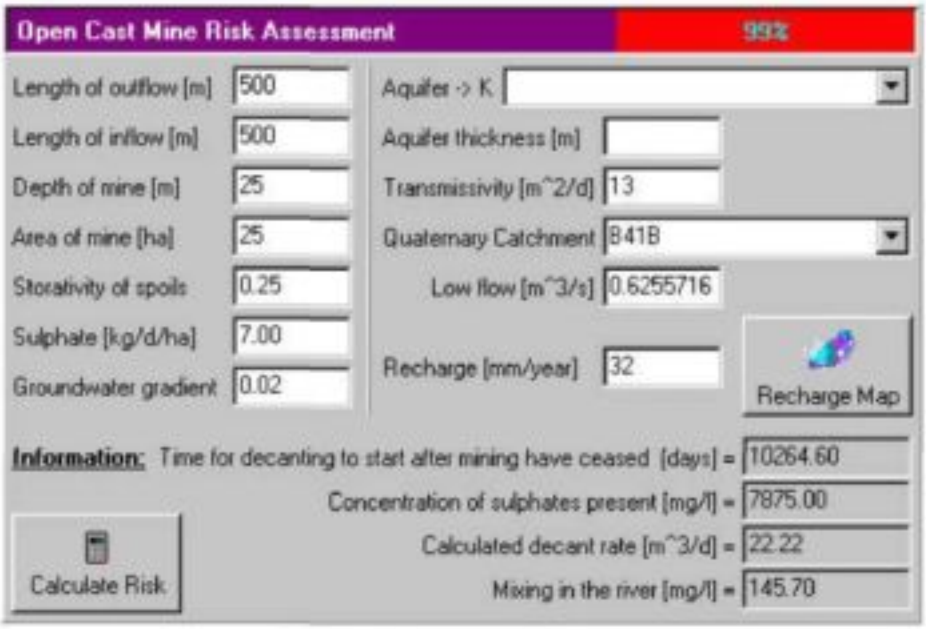


Figure 5-6. Open cast mine risk assessment

CHAPTER 6

Health Risk Assessment

6.1 INTRODUCTION

A health risk assessment is the process or method of determining if an activity (man-made or natural) will negatively impact human health. As such a health risk assessment is a decision making tool. It provides the analytical support for decisions that protect public health (Schwab and Genthe, 1998).

The scope and nature of a health risk assessment varies – from broadly based scientific conclusions about arsenic affecting the whole nation to site-specific findings concerning the same chemical in the local water supply.

Lung disease, cancers and chronic poisoning are some of the hazards associated with chemicals and dusts in projects ranging from the application of agricultural chemicals to quarrying and mining. Exposure may occur at both places of occupation and residence through unregulated emissions to various media (in this case groundwater), or through the inappropriate use of machinery, for example leaking engines. In some instances, projects with obvious health benefits may also have unintentional health impacts. Water supply projects, for example, will often reduce the occurrence of diseases such as diarrhea and cholera, but if incorrectly sited and managed, may induce other diseases. For example when a wellfield is located close to fertilized crops, the nitrate in the groundwater can cause methyglobinemia, a type of anaemia, which can be fatal for infants.

A groundwater health risk assessment can be defined as a qualitative or quantitative process to characterise the probability of adverse health effects associated with measured or predicted levels of hazardous agents in groundwater. Once a contaminant is released into the groundwater, its resultant concentrations found in the human body is dependent upon the physical and chemical properties of both the contaminant and the groundwater. In addition the concentrations found in a human are subject to the person's exposure to groundwater. Exposure is defined by the frequency, magnitude and duration of contact with the contaminant. Frequency refers to whether a person is exposed daily or just occasionally. The magnitude refers to the amount of exposure; occupational exposure will be greater than community exposure. The duration refers to whether any single exposure episode may last for minutes, hours, days or years. Once the contaminant is inside the body it may be further transformed via metabolism or detoxification. The ability to transform chemicals varies. Children, the elderly and those with chronic conditions, for instance, react differently to the same dose than the average, healthy middle-aged adult (Schwab and Genthe, 1998). The impact of contaminants for the various scenarios are characterised in a health risk assessment.

A health risk assessment consists of problem formulation (hazard identification), analysis and risk charaterisation. Figure 6-1 is a summary of these.

6.2 DATA REQUIREMENTS

Unlike groundwater risk assessments, there is a large difference in the methods used to calculate the three tiers of health risks. However, they will all use fuzzy logic rules to some degree.

According to the definition of a rapid assessment, low levels of confidence are attached to the results and no intensive field investigations are necessary. Therefore the data requirements are limited and the assessment will rely heavily on data stored in the DT database. One of the main differences between the rapid and intermediate assessment is the quality of data required for the assessments. Table 6.1 is a summary of the data required for the assessments.

Groundwater health risk assessment

Problem formulation

Hazard identification establishes whether exposure to a chemical, radioactive or microbiological agent can cause harm. This step determines whether the risk assessment should be continued or abandoned. Once a health hazard has been identified, the remainder of the process encompasses the description of the properties of the hazardous agent, and the identification of its health effects.

Questions to be asked: What is the problem? Why is it a problem? How was the problem first recognised? What are the effects on human health? How urgent is the need for action?

Analysis

Exposure Assessment

Establishes the intensity, frequency and duration of human contact with a contaminant. To determine exposure, it is necessary to combine an estimation of environmental concentrations of the hazards with demographic or behavioral descriptions of the exposed

Dose-response Assessment

Characterises the relationship between the dose of a hazardous agent (ie the amount of pollutant taken into the body through breathing, ingestion and skin contact) and incidence of an adverse effect in the exposed population.

Risk characterisation

Provides an indication of the incidence of the health effect under the conditions of exposure described in the exposure assessment and identified dose-response relationship. Risk characterisation should include information that is useful to both stakeholders and risk managers.

Questions to be asked: What is the nature and likelihood of the health risk? Which individuals or groups are at risk? How severe are the anticipated adverse effects? What scientific evidence supports the conclusions about the risk? How strong is the evidence?

Figure 6-1. Methodology for a groundwater health risk assessment

Table 6.1: Data required health risk assessments and potential data sources

Data Required	Potential data sources		
	Rapid	**Intermediate**	**Comprehensive**
Chemical	User to choose from list in DT	User to choose from list in DT	User to choose from list in DT
• Concentration	-	Laboratory analyses	Laboratory analyses
• Cancer potency factor	-	DT database	DT database
• Reference dose	-	DT database	DT database
Microbiological agent	User to choose from list in DT[1]	User to choose from list in DT database[2]	User to choose from list in DT database[2]
• Number of organisms	-	Laboratory analyses	Laboratory analyses
• α and β or r	-	DT database[2]	DT database[2]
Radioactive element	User to choose from list in DT	User to choose from list in DT database	User to choose from list in DT database
• Concentration	-	Laboratory analyses	Laboratory analyses
• Risk coefficient	-	DT database	DT database
Exposure duration	User to choose from values in DT[3]	User	User
Population subgroup	User to choose from values in DT[3]	-	-
Population size	User to choose from values in DT[3]	User to obtain from local authorities	Field survey
Average intake rate	-	Average values documented in DT database[4]	Field data
Body weight	-	Average values documented in DT database[4]	Field data
Life time age of person	-	Average values documented in DT database[4]	Field data

[1]Refer to Appendix D, Section D1.
[2]Refer to Appendix D, Section D2.
[3]Refer to Table 6.2.
[4]Refer to Appendix D, Section D3.

6.3 ASSUMPTIONS AND LIMITATIONS

The assumptions and limitations for the various assessments differ and will therefore be discussed separately.

Rapid Assessment

- As this is a rapid assessment, no analyses are needed, therefore the concentrations (or dose) of the chemicals, radioactive and microbial agents are not included in the assessment.
- In addition exact exposure durations are not included in the calculations.
- The exposure pathway (oral, dermal and inhalation) is not taken into account.

Intermediate Assessment

- It is assumed that an adult weighs 70 kg,
- It is assumed that a child weighs 10 kg,
- It is assumed that a person drinks 2 liters of water a day,
- It is assumed that a person inhales 20 m^3 of air a day,
- It is assumed that the average lifetime of human is 70 years.

General – applicable to all assessments

- Only direct exposure (oral, dermal or inhalation) to groundwater are considered. Indirect pathways are not taken into account, for example eating foods irrigated with contaminated groundwater.
- Due to the shortage of information concerning membership functions, the functions will either be cosine or specific values will be assigned to specific conditions.
- If the pollutant is not carcinogenic, radiogenic, toxic or causes infection then health risks are considered to be zero.
- If the exposure to a pollutant is zero, then the health risks are considered to be zero.
- Only the carcinogenic effects of radioactive elements are considered.
- The radiogenic risk coefficients used in the intermediate and comprehensive assessments do not include the contribution of daughter products.

6.4 METHODOLOGY

6.4.1 General

Experts in the field highlight the following components as important when performing a health risk assessment:

- Toxicity of the contaminant: When exposed to toxic chemicals there are numerous health effects that vary from mild headaches to death, all of which need to be taken into account in a risk assessment.
- Carcinogeneity of a contaminant: Cancers traced to direct, involuntary exposure to environmental pollution are estimated to constitute about 2% of all cancer risks (Doll and Peto, 1981). However exposure to certain chemicals can cause some form of cancer and therefore the carcinogeneity of a chemical needs to be taken into account when conducting a health risk assessment.
- Possibility of infection: Allows the user to obtain an idea of the risks involved in human exposure to a variety of bacteria, viruses and protozoa. The risk of infection is 10 to

1000 times less for the bacteria than the viruses and protozoa at similar levels of exposure.

- Radiation exposure can result in delayed effects such as cancer. Some of the cancers associated with radiation are: leukemia, esophagus, stomach, colon, liver, lung, breast, ovary, urinary tract and multiple myeloma.
- Exposure to a contaminant: This establishes whether exposure to a chemical or microbiological agent can cause harm. To determine exposure, it is necessary to combine an estimation of environmental concentrations of the hazards with demographic or behavioral descriptions of the exposed population.
- Population exposed to a contaminant: The population is composed of groups who differ in their vulnerability to health hazards. For example babies are more susceptible to infection because of their lack of immunity.
- Size of exposed population: The seriousness of health risks does not only depend on the hazardous agents but also on the size of the population affected. In general the bigger the population, the more cost and effort needed to treat the resulting health impacts.

The fuzzy logic methodology will once again be used in the risk assessments, however the decision rules and membership functions will differ for the rapid assessment. Therefore this assessment will be discussed separately.

The decision rules and membership functions were determined with the assistance of one of the expert health risk assessors, Bettina Genthe from the CSIR.

6.4.2 Rapid groundwater health risk assessment

The methodology is the same as discussed in Section 3.2. Therefore it is once again necessary to set up decision rules and membership functions. The membership functions are listed in Table 6.2, and the decision rules are documented in Appendix F, Section F3.

Table 6.2: Membership functions/values for the rapid health risk assessment

Rapid Health Risk Assessment Membership Functions			
Toxic		Infection	
Range[1]	Membership value	Range[2]	Membership value
Death	0.0	Death	0.0

Effects common	0.25	Effects common	0.25
Long-term effects	0.4	Long-term effects	0.4
Few effects	0.8	Few effects	0.8
Carcinogen[3]		**Exposure duration**	
Range[4]	**Membership value**	**Exposure**	**Membership value**
A: known human carcinogen	0.0	Contamination may be seconds, minutes or hours	0.9
B1: probable human carcinogen (limited data)	0.25	Contamination occurs at intermittent periods < 2 years	0.6
B2: probable human carcinogen (inadequate data)	0.25	Contamination > 90 days and < 2 years	0.6
C: possible human carcinogen	0.5	Contamination occurs at intermittent periods > 2 year	0.3
D: Not classified as a human carcinogen	0.75	Continuous contamination > 2 years	0.0
E: Evidence that not a human carcinogen	1.0		
Population subgroups		**Size of exposed population**	
Subgroup	**Membership values**	**Number of people**	**Membership values**
Children under the age of 2 years	0.0	> 500000	0.0
Elderly over the age of 60 years	0.0	100000 – 500000	0.05
Adults with chronic conditions	0.0	10000 – 100000	0.4
Adults between 30 and 60 years	0.5	5000 – 10000	0.65
Children between 2 and 20 years	0.5	< 5000	0.9
Adults between 20 and 30 years	0.9		

[1]Taken from water quality guidelines, Appendix C (Section C2) contains more information.
[2] See Appendix D, Section D4 for more information.
[3]According to the US EPA (1994) all radioactive elements are classified as known human carcinogens (A).
[4]Taken from US EPA Classification of Carcinogens (EPA, 2000).

The rapid groundwater health risk can be determined using Equation 3.1.

6.4.3 Intermediate and comprehensive groundwater health risk assessment

The main difference between intermediate and comprehensive assessments is the quality of data. Both intermediate and comprehensive assessments involve field work and are divided into a:

- toxic assessment,
- carcinogenic assessment,
- microbiological assessment and
- radiogenic assessment.

The intermediate assessment only requires concentrations of the contaminant to be determined, while the comprehensive assessment requires information concerning the affected population and their lifestyle.

6.4.3.1 Toxic and Carcinogenic Assessment

Toxic and carcinogenic assessments take into account the routes of exposure as documented in Table 6.3.

Table 6.3: Exposure pathways considered in groundwater driven risk assessments (Maxwell *et al.*, 1998)

Routes of exposure	Groundwater exposure pathway
Ingestion	Drinking groundwater
Inhalation	Inhalation of contaminant transferred from water to vapor in air
Dermal sorption	Sorption through skin in baths and showers

Before calculating the risks associated with both these assessments, the total dose, average daily dose and lifetime average dose have to be defined. The equations used to define risks associated with human exposure to contaminated groundwater are generally based on those specified in the EPA "Risk Assessment Guidance for Superfund" (EPA, 1989).

For each pathway, the total dose that will reach a human has to be calculated. The total dose is defined as:

$$\text{Dose} = C \times IR \times ED$$

where

Dose = Total dose

C = Maximum concentration

IR = Average intake rate

ED = Exposure duration

The average daily dose is determined by dividing an estimate of the total dose accrued during the exposure duration from a pathway by an averaging time or an expected lifetime:

$$ADD = \frac{Dose}{BW \times ED}$$

where

BW = Average body weight over exposure period

Carcinogenic risk assessments are determined over a human's lifetime. Therefore the lifetime average daily dose (LADD) is calculated as:

$$LADD = \frac{\text{Total dose}}{BW \times \text{lifetime}}$$

The carcinogenic risk calculation is based on a Poisson model:

$$Risk = 1 - e^{-LADD \times CPF} \approx LADD \times CPF$$

where CPF is the cancer potency factor. The potency factor is the slope of the percentage of animals developing cancer versus the dosage level of a particular chemical. The slope of this curve is then extrapolated to the low doses expected to be encountered by humans who may be exposed to the same chemical. Because of the complex uncertainty involved with calculating the cancer potency, these values are obtained from the IRIS (EPA, 1988) database and included in the database of the DT.

The toxic risk is calculated as:

$$Risk = \frac{ADD}{RfD}$$

where

RfD = Reference dose

The reference dose is an estimate of daily exposure to the population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime. These values are documented in the IRIS database (EPA, 1988) and will be included in the DT database.

6.4.3.2 Microbiological Assessment

Microbiological contamination of water is the largest and most immediate health hazard (Genthe and Rodda, 1999). There are two models used to determine the probability or risk of infection from pathogens (Rose and Gerba, 1991):

- The single-hit exponential model

$$\text{Risk} = 1 - e^{(-rN)}$$

- The beta-distributed model

$$\text{Risk} = 1 - \left[1 + \left(\frac{N}{\beta}\right)\right]^{-\alpha}$$

where risk refers to risk or probability of infection and

N = Exposure (number of organisms).

α,β & r = Parameters characterised by dose-response curves.

In most cases the beta-distributed model is the most appropriate, however in the case of Giardia lamblia and Cryptosporidium parvum the single-hit exponential model is to be used (Rose and Gerba, 1991). Values for α,β and r are included in the database of the DT (Appendix D, Section D2).

6.4.3.3 Radiogenic assessment

The methodology followed for the radiogenic assessment is based on that documented by the US EPA (1994(a)). A radiogenic assessment is divided into two types of assessments:

- Mortality risk: the age- and gender-specific or total risk of people dying from radiation induced cancers.
- Morbidity risk: the age- and gender-specific or total incidence of radiation induced cancers.

Both risks will be calculated. However, the higher of the risks will be used in the decision rules. In most cases this will be the morbidity risk.

The risk calculations are based on a risk coefficient (r) developed by the US EPA (1994(a)). The risk coefficients represent an estimated radiogenic cancer risk, reflecting the age and gender distribution. The coefficients can be used for short-term and long-term exposures.

For a selected exposure scenario, the calculated risk involves the multiplication of the applicable risk coefficient by the dose:

$$\text{Risk} = r \times \text{Dose}$$

The above equation is correct for inhalation and ingestion. However for submersion the risk coefficient is expressed not only in terms of becquerel but also in terms of volume and time, therefore the submersion risk is calculated as:

$$\text{Risk} = r \times C \times ED$$

6.4.3.4 Intermediate and Comprehensive Health Risk Calculations

The intermediate and comprehensive assessments take into account the carcinogenic, toxic, microbial and radiogenic risks together with the size of the population. The membership functions/values are listed in Table 6.4 and the decision rules are documented in Appendix F, Section F3. The final risk is calculated using Equation 3.1.

6.5 EXAMPLE

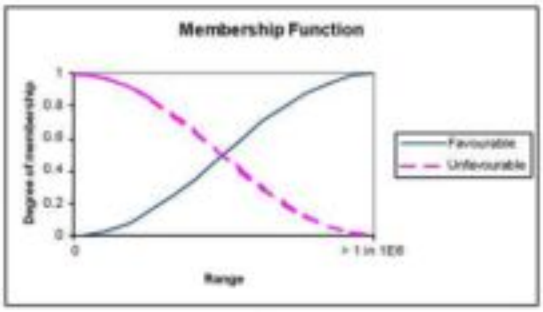
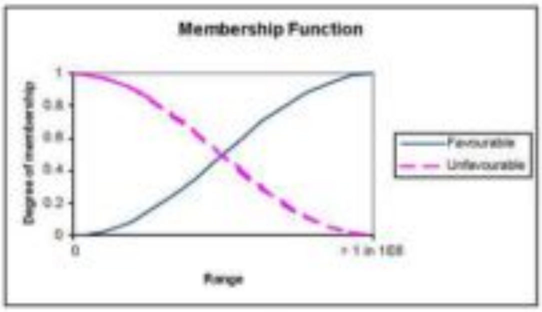
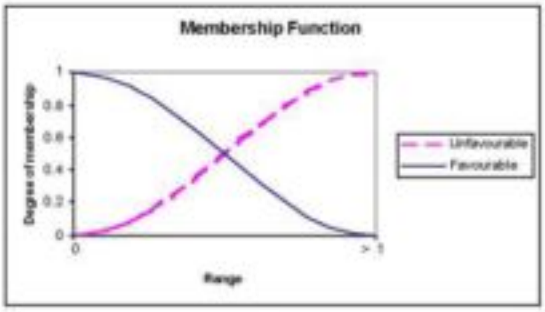
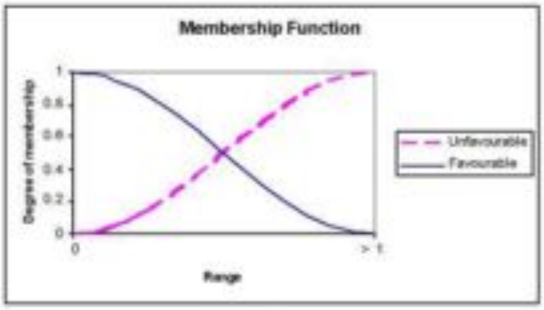
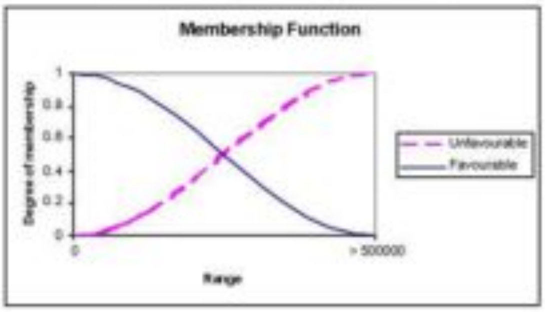
Consider a rapid assessment example of water supply boreholes situated close to an industrial area. Most of the people dependent on the boreholes are between 20 and 30 years old, and young children under the age of 2 years. The size of the population is approximately 20 000 people.

Benzene has been found in the groundwater. Benzene is a colourless liquid with a sweet odour. It evaporates into the air very quickly and dissolves slightly in water. Long-term exposure to high levels of benzene in the air can cause leukaemia, cancer of the blood-forming organs. Breathing very high levels of benzene can result in death, while high levels can cause drowsiness, dizziness, rapid heart rate, headaches, tremors, confusion, and unconsciousness. Eating or drinking foods containing high levels of benzene can cause vomiting, irritation of the stomach, dizziness, sleepiness, convulsions, rapid heart rate, and death. It is clear that benzene is both a carcinogen and a toxin.

Two scenarios were calculated: one for the children under the age of 2 years and one for adults between the ages of 20 and 30 years. The results of both assessments produce a risk of 99%.

Table 6.4: Membership functions/values for the intermediate and comprehensive health risk assessment

Membership Functions[1]

<table>
<tr><td colspan="2">Radiation[2]

</td><td colspan="2">Carcinogen[3]

</td></tr>
<tr><td>Unfavourable limit</td><td>Favourable limit</td><td>Unfavourable limit</td><td>Favourable limit</td></tr>
<tr><td>1 in 10^2 - 10^3</td><td>> 1 in 10^6</td><td>1 in 10^2 - 10^3</td><td>>1 in 10^6</td></tr>
<tr><td colspan="2">Infection

</td><td colspan="2">Toxin

</td></tr>
<tr><td>Unfavourable limit</td><td>Favourable limit</td><td>Unfavourable limit</td><td>Favourable limit</td></tr>
<tr><td>≥ 1</td><td>< 0.2</td><td>≥ 1</td><td>< 0.2</td></tr>
<tr><td colspan="4">Size of exposed population

</td></tr>
<tr><td colspan="2">Unfavourable limit</td><td colspan="2">Favourable limit</td></tr>
<tr><td colspan="2">> 500000</td><td colspan="2">< 5000</td></tr>
</table>

[1]Membership functions are documented in Appendix D, Section D4.
[2]1 in 10^x means one person out of every 10^x people exposed has the chance of developing cancer or dying from cancer depending on morbidity or mortality risk respectively.
[3]1 in 10^x means one person out of every 10^x people exposed has the chance of developing cancer.

Figure 6-2 shows the input screen and results of one of the scenarios. It must be noted that in the rapid health risk assessment the user can only select one toxin, one microbiological agent and one carcinogen.

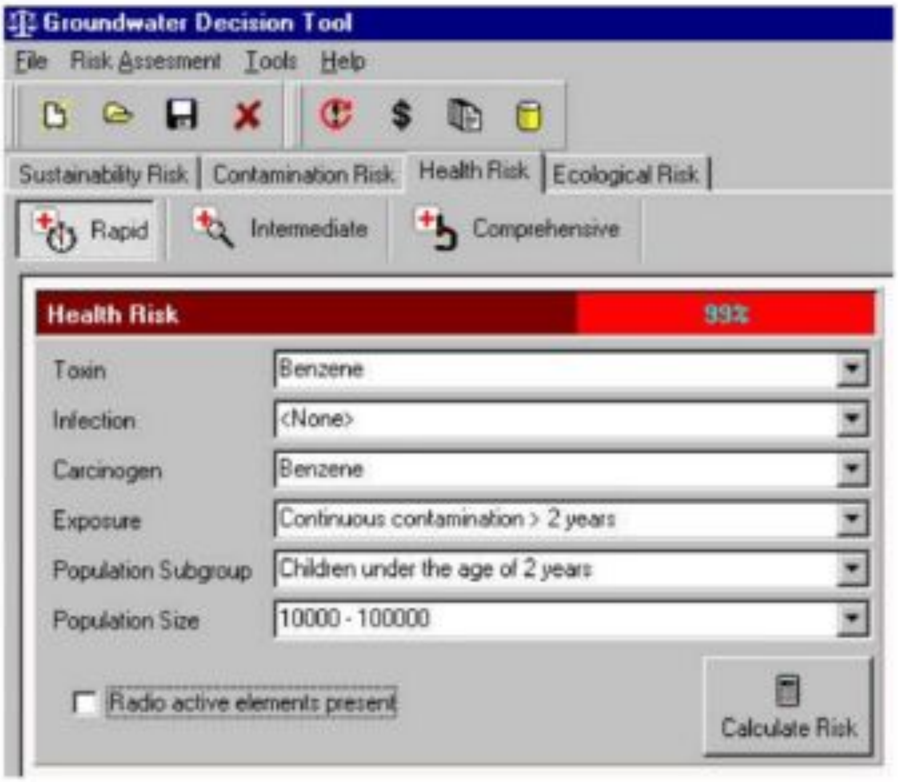


Figure 6-2. Results of rapid health risk assessment

The intermediate and comprehensive assessments are similar. Therefore, only an example of an intermediate risk will be conducted here. Take for example a hypothetical water supply borehole once again situated close to an industrial area. Analyses of the groundwater indicates the presence of hexachlorobenzene. Hexachlorobenzene was widely used as a pesticide to protect the seeds of onions, sorghum, wheat, and other grains against fungus until 1965. It was also used to make fireworks, ammunition, and synthetic rubber. Studies in animals show that ingesting hexachlorobenzene for a long time can damage the liver, thyroid, nervous system, bones, kidneys, blood, and immune and endocrine systems. The immune system of rats that breathed hexachlorobenzene for a few weeks was harmed. There is no strong evidence that it causes cancer in people.

In addition there are many pit latrines in the area. Further analyses of the groundwater indicate that *Shigella dysenteriae* is also present. *Shigella dysenteriae* causes acute disease of the large and small intestine, diarrhea, fever, nausea, and sometimes toxemia, vomiting, cramps and tenesmus. The infections have up to a 20% fatality rate in hospitalised patients. A person can be exposed to *Shigella dysenteriae by* direct or indirect fecal-oral transmission from a patient or carrier. Poor hygiene practices spread infection to people by direct physical contact or indirectly by contaminating food and water.

The data used for the risk assessment are listed in Table 6.5. The results of the risk assessment can be seen in Figure 6-3.

Table 6.5: Data required for an intermediate health risk assessment

Data Required	Value
Exposure pathway	Ingestion
Population size	10 000 – 100 000
Carcinogen • Concentration (mg/L)	Hexachlorobenzene 0.03
Toxin • Concentration (mg/L)	Hexachlorobenzene 0.03
Radiation	-
Micro-organism • Number of organisms	Shigella dysenteriae 200
Exposure duration (days)	360

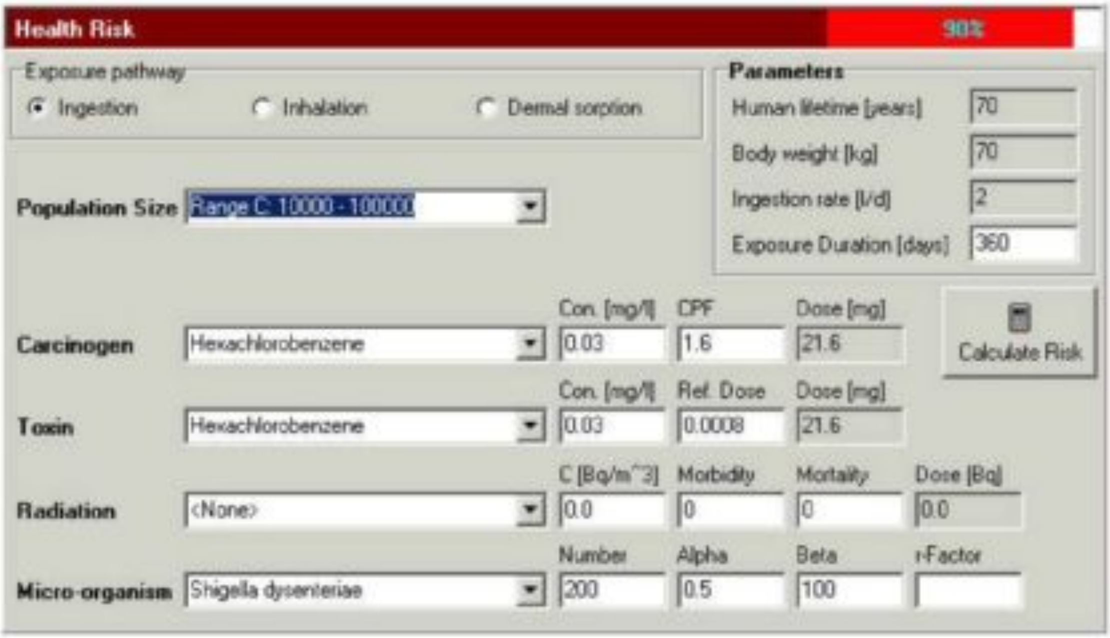


Figure 6-3. Results of intermediate health risk assessment

For a comprehensive assessment the user can choose more than one carcinogen, toxin, radioactive element and micro-organism. The DT will work out the risk taking all selected hazardous elements into account. The exact size of the population under consideration must also be provided.

CHAPTER 7

Ecological Risk Assessment

7.1 INTRODUCTION

7.1.1 Preamble

The South African National Water Act (Act No 36, 1998) is based on a number of principles one of which is: *the quantity, quality and reliability of water required to maintain the ecological functions on which humans depend shall be reserved so that the human use of water does not individually or cumulatively compromise the long term sustainability of aquatic and associated ecosystems* (Scott and Le Maitre, 1998). The Water Act does however focus on aquatic ecosystems and therefore this Chapter will focus on the risk of negative impacts of groundwater (quantity and quality) on aquatic ecosystems.

Aquatic ecosystems are defined as the abiotic (physical and chemical) and biotic components, habitats and ecological processes contained within rivers and their riparian zones, reservoirs, lakes and wetlands and their fringing vegetation (DWAF, 1996).

Ecological risk assessments differ from health risk assessments in several significant ways. For ecosystems, the risk assessment methodology must consider effects beyond just individual organisms or a single species. No set of ecological values and tolerances applies to all of the various types of ecosystems. With ecosystems some sites and types are more valuable and vulnerable than others. Accommodating these factors complicates ecological risk assessments and renders them more subjective. Unfortunately, there are limited data available concerning South African aquatic ecosystems. This investigation will therefore only consider factors such as groundwater-surface water interactions, groundwater-vegetation dependence, the uniqueness of the ecosystem, groundwater base flow versus abstraction and South African water quality guidelines. These will be used as indicators of the health of the entire aquatic ecosystem. A summary of the general steps in a ecological risk assessment is shown in Figure 7-1.

Groundwater ecological risk assessment

GROUNDWATER QUANTITY

Problem formulation

Establishes the impacts of groundwater on ecosystems taking into account (1) difference between groundwater levels and root depth, (2) type of dependence, (3) surface-groundwater interactions, (4) the groundwater component of base flow and (5) the uniqueness of the ecosystem.

If there is no interaction between groundwater and surface water both the groundwater quantity and quality assessments can be abandoned.

Analysis

Determine groundwater-surface water interactions (including base flow), determine dependence of vegetation on groundwater. Estimate the depths of root systems for vegetation. Determine the amount of water needed to sustain vegetation. Take into account the uniqueness of the ecosystems under investigation.

GROUNDWATER QUALITY

Problem formulation

Establishes whether groundwater quality has an impact on ecosystems and if so, what the impact is. If there is no impact the assessment can be abandoned.

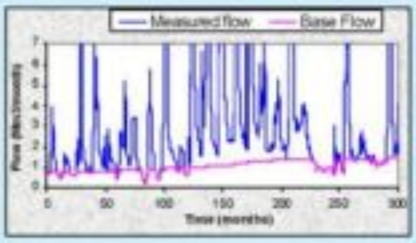
Questions to be asked: Does groundwater quality have an impact on ecosystems? If so what is the impact of groundwater quality on ecosystems?

Analysis

Estimate the water quality required for aquatic ecosystems.

Determine the quality of groundwater.

Determine the impact of changes in groundwater quality on ecosystems taking into account the frequency and duration of changes.



Risk characterisation

Provides an indication of the risks as a result of changes in groundwater levels or quality on ecosystems under the conditions discussed under analysis.

Questions to be asked: What is the nature and likelihood of groundwater impacting ecosystems? How severe are the anticipated adverse effects? What scientific evidence supports the conclusions about the risk?

Figure 7-1. Diagram summarising the ecological risk assessment methodology

7.1.2 Aspects taken into account when determining ecological risks

7.1.2.1 Groundwater-surface water interactions

The first and important aspect to take into account is groundwater-surface water interaction. This has to be determined in order to determine if groundwater plays a role in the sustainability of the aquatic ecosystem under investigation. Scott and Le Maitre (1998) define a number of types of interactions between groundwater and surface water. These are summarised into the following broad categories:

- Influent: The groundwater level is lower than the surface water level, and therefore surface water is recharging groundwater.
- Effluent: The groundwater level is higher than surface water level, and therefore groundwater is recharging surface water.
- Intermittent: The groundwater level is higher than the bed of the surface water body, but depending on the elevation of the water level, groundwater may recharge the surface water body or the surface water may recharge groundwater.
- Detached: The groundwater level is below the surface water level and the two do not influence each other.

These interactions are depicted in Figure 7-2. If the surface water body and the groundwater system are detached or the surface water body is influent then the ecological risks due to groundwater are zero.

The amount of base flow from groundwater entering a surface water body has direct impacts on the aquatic ecosystems present in and surrounding the water body.

7.1.2.2 Dependency of vegetation on groundwater

The degree of dependence of vegetation on groundwater as a source of water and survival is important when determining ecological risks. The dependency on groundwater can be classified as (Scott and Le Maitre, 1998):

- Obligatory phreatophytes obtain their water supply from the saturated zone and are most vulnerable to impacts caused by groundwater exploitation, or some other case of reduced groundwater levels. These phreatophytes can be sub-divided according to Hatton and Evans (1998) into the following classes:

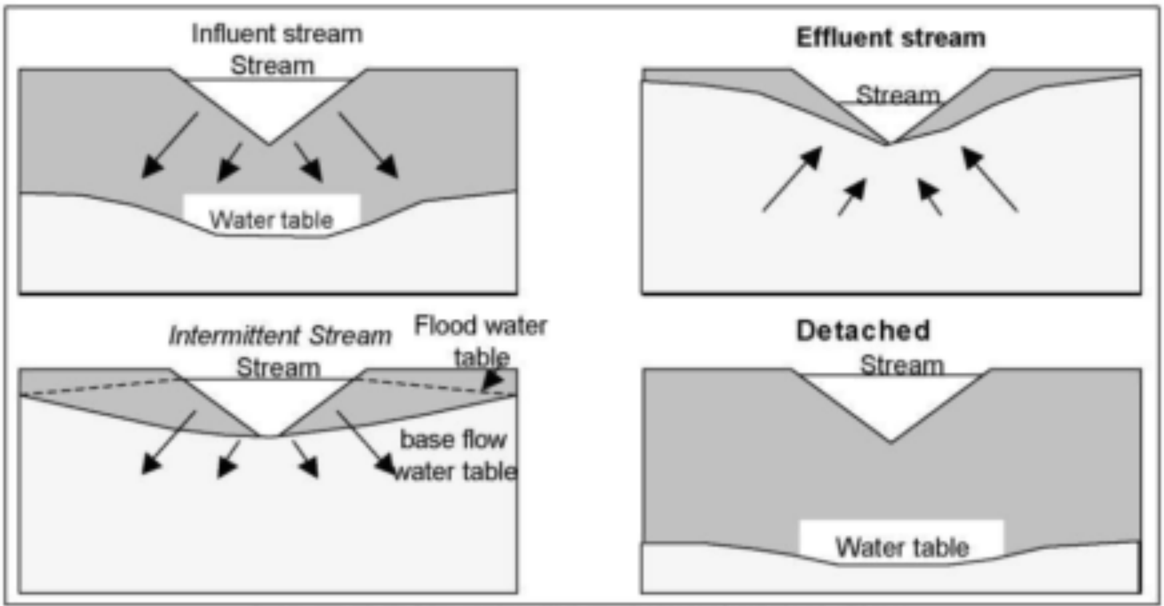


Figure 7-2. Groundwater-surface water interactions

- Entirely – The ecosystem is entirely dependent on groundwater, and were groundwater to diminish or be modified only slightly, either below a threshold like the ground surface or such that a surface water body stops flowing, the ecosystem will be destroyed. Examples of such ecosystems can be related to springs, permanent lakes and groundwater discharges into the saline bodies such as the sea.
- Highly – Moderate changes to groundwater discharge or water tables would lead to substantial decreases in the extent and health of the ecosystem. An example of such an ecosystem is a swamp.
- Proportionally – For a number of systems it is likely that a unit change in the amount of groundwater will result in a proportional change in the health of that ecosystem. In other words, if the groundwater discharge is halved, one might expect the same diminution of the ecosystem. An example of such an ecosystem can be the river plains of a perennial river.

- Facultative phreatophytes exploit groundwater without being dependent on it for survival.
- Vegetation that is not dependent on groundwater.

7.1.2.3 Depth of root system

The depths of plant root systems are highly variable and systematic studies are rare. Although deep roots may only comprise a small fraction of the rooting system they may be critical for plant survival, a few deep roots can even sustain large trees. Studies of diurnal and seasonal water relations show that species able to maintain root systems in contact with groundwater tables have high transpiration rates and show little seasonal variation in water stress (Scott and Le Maitre, 1998).

7.1.2.4 Uniqueness of ecosystem

The seriousness of ecological risks does not only depend on the hazard but also the uniqueness of the ecosystem being affected. Ecosystems are therefore classified according to endangered, sensitive indigenous, indigenous and alien species. Even though the uniqueness of the system will be impacted by both water quantity and quality changes, for the sake of convenience, it will be included in the quantity assessment.

7.1.2.5 Change in water quality

Change in groundwater quantities and groundwater contamination can have a direct impact on aquatic ecosystems. The changes in water quality can cause the following (DWAF, 1996):

- Chronic effects – This is defined as that concentration or level of a constituent at which there is expected to be a significant probability of measurable chronic effects in up to 5% of species in the aquatic ecosystem.
- Acute effects – This is defined as that concentration or level of a constituent above which there is expected to be a significant probability of acute toxic effects in up to 5% of the species in the aquatic ecosystem. If an acute effect persists for even a short while, or occurs at too high a frequency, it can quickly cause the disappearance of sensitive species.

According to the South African Water Quality Guidelines (DWAF, 1996) there are 4 categories of constituents that effect aquatic ecosystems:

- Toxic constituents – Seldom occur in high concentrations in unimpacted systems. Examples are inorganics (Al, As, Cd, Cu F^-, Hg, Mn and NH_4^+) and organics (phenol and atrazine).
- System variables (for example pH) – Regulate essential ecosystem processes such as spawning and migration. Changes in the amplitude, frequency and duration of natural seasonal cycles may cause severe disruptions to ecological and physiological functions of organisms.
- Non-toxic inorganic constituents – May cause toxic effects at extreme concentrations, for example total dissolved solids (TDS) and total suspended solids (TSS).
- Nutrients – Are generally not toxic, but can stimulate eutrophication if present in excessive quantities.

7.1.2.6 Duration of exposure to contamination

If the contamination is a once-off spill the impacts will most probably be smaller than those of a continuous source.

7.2 DATA REQUIREMENTS

As with all risk assessments, a fuzzy logic system will be used to determine the ecological risks. The fuzzy rules will remain the same for all tiers of the risk assessment, however the quality of data as well as methods used to determine groundwater base flow vary. Table 7.1 and Table 7.2 is a summary of the data requirements and potential data sources.

Table 7.1: Data required for the quantity ecological risk assessment and potential data sources

Data required	Potential data sources		
	Rapid	**Intermediate**	**Comprehensive**
Water quantity			
Perennial/Non-perennial	User	User or DT	User
Groundwater level	User to estimate	User from field data	User from field data
Vegetation type to determine root depth	User to give vegetation type corresponding root depths in DT database[1]	User to give vegetation type corresponding root depths in DT database[1]	User from field data
Groundwater-surface water interaction	User from DT database[2]	User from field data	User from field data
Type of dependence	User	User	User to determine from field data
Groundwater component of base flow			
• Primary catchment	User[3]	-	-
• Quaternary catchment	-	User	
• Total monthly flows			User
• In flow stream requirements	-	-	User
Influence of abstraction on groundwater base flow			
• Abstraction rate in borehole(s)[4]	User	User	User
Uniqueness of ecosystem	User	User	User

[1]See Appendix E, Section E1. [2]See Appendix E, Section E2. [3]See Appendix E, Section E3.
[4]Only one borehole is included in the rapid assessment.

Table 7.2: Data required for the quality ecological risk assessment and potential data sources

Data required	Potential data sources		
	Rapid	**Intermediate**	**Comprehensive**

Water quality			
Toxin	User	User	User
• Aquatic ecosystem guidelines	DT database	DT database	DT database
• Concentration	-	Laboratory analyses	Laboratory analyses
• Injection rate	-	-	User
• Duration of injection	User[1]	User[1]	User[1]
Hydraulic conductivity	-	User OR	User
• Aquifer type	-	DT database[2]	
Porosity	-	User OR	
• Aquifer type	-	DT database[3]	
Groundwater gradient	-	User	User[4]
Dispersivity	-	User[5]	User[6]
• Borehole position, concentration & time	-	-	User[6]
Source			
• Source position	-	-	User
• Distance between source and riparian zone	-	User	User
Evaluation time	-	User	User

[1]See Table 7.4. [2]See Appendix C, Section C7 [3]See Appendix C, Section C8.
[4]Can use Tripol software to calculate groundwater gradient.
[5]User to enter dispersivity value or the DT will use the distance between source and riparian zone to calculate dispersivity.
[6]User to enter dispersivity value or user can enter a borehole position, concentration and time. The DT will then calculate dispersivity.

7.3 ASSUMPTIONS AND LIMITATIONS

Rapid Assessment

- Only abstraction borehole closest to the aquatic ecosystem is taken into account.
- Literature, maps, rules of thumb and elementary calculations are used to determine the membership functions and decision rules.
- Does not take concentrations into consideration.

Intermediate Assessment

- When calculating the concentration of contamination at any point in the aquifer only advective-dispersive processes are taken into account.

Comprehensive Assessment

- When calculating the concentration of contamination at any point in the aquifer only advective-dispersive processes are taken into account. The source can however be a slug or a continuous source.

General – applicable to all assessments

- Surface water levels and flow conditions in surface water bodies are not taken into account.

- In many cases it is necessary for an ecosystem to experience some stress. However, reducing the availability of groundwater too much may result in a gradual decrease in the health of the ecosystem. On the other hand, an abundance of groundwater can also be harmful to ecosystems. For example the roots of certain plants can be drowned. These changes in groundwater levels are not taken into account in the DT.
- Fish and other invertebrates are indirectly taken into account by applying aquatic ecosystem guidelines and considering the groundwater component of base flow.
- If a river is non-perennial it is assumed that all risks are zero.
- Only toxic constituents will be taken in account in the risk assessment process. However other factors (such as system variables and nutrients) will be discussed and taken into account in protecting and remediating a resource. This will be discussed in more detail in Chapter 8.
- The distance of a borehole from a surface water body is not taken into account.

7.4 METHODOLOGY

The methodology to assess the ecological risk is the same as discussed in Section 3.2. Therefore, it is once again necessary to set up decision rules and membership functions, which are the same for all tiers of the assessment. These rules and functions have been discussed with Gerrit van Tonder (Institute for Groundwater Studies), Christine Colvin (CSIR) and Dave Le Maitre (CSIR) all of whom have conducted research on ecosystem-groundwater interactions. The quality of data and calculations for the various tiers of assessments differ. The calculations needed for the risk assessment are summarised in Table 7.3. The membership functions for the decision rules are documented in Table 7.4.

Table 7.3: Calculations for aquatic ecosystems risk assessment

<table>
<tr><th colspan="4">QUANTITY ASSESSMENT</th></tr>
<tr><th>Data required</th><th>Rapid Assessment</th><th>Intermediate Assessment</th><th>Comprehensive Assessment</th></tr>
<tr><td>The following is required to determine base flow:
• The primary catchment for the rapid assessment.
• The quaternary catchment for the intermediate assessment.
• Total monthly flow and inflow stream requirements (IFRs) for the comprehensive assessment.

The base flow value (and IFR) is compared to the amount of groundwater being abstracted (see Table 7.3).

It is important to note that if the percentage noflow is not equal to zero then there are no impacts of groundwater on aquatic ecosystems and the risk is zero.</td><td>The natural base flow is determined from the Vegter and Pitman (1996) primary catchment values (See Appendix E, Section E3).

Only one abstraction borehole is taken into account</td><td>The base flow is determined using the SARES program by Hughes (1999).

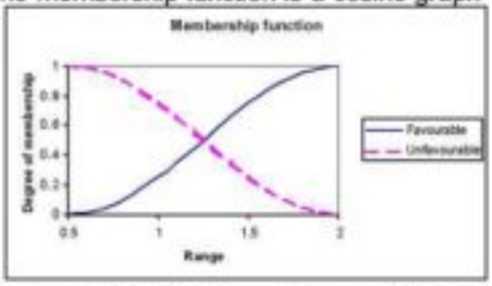
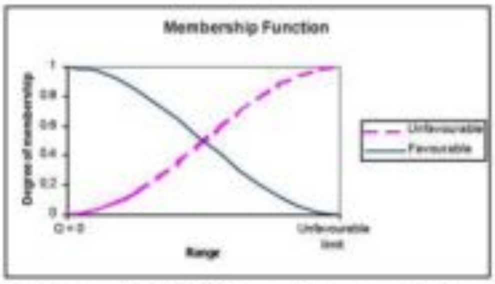
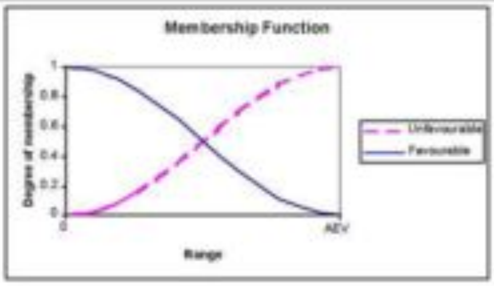
All abstraction boreholes are taken into account.</td><td>Base flow must be determined using one of the existing methodologies for example Van Tonder (2001).

All abstraction boreholes are taken into account.</td></tr>
<tr><td>To estimate the depths of root systems the following data are needed:
• Types of vegetation for the rapid and intermediate assessments.
• Depths of root system from field investigations for comprehensive assessment.

The water table in the riparian zone is compared to the root depths. If there is more than a 2 m difference it is accepted that the vegetation under investigation is not dependent on groundwater, if there is less than 0.5 m difference it is accepted that the vegetation is totally dependent on</td><td colspan="2">The vegetation type is used to estimate the average root depth (See Appendix E, Section E1).
OR
The user can enter the root depth.</td><td>A field investigation is necessary to determine the types of plants potentially dependent on groundwater together with their associated root depths.</td></tr>
</table>

groundwater.			
QUALITY ASSESSMENT			
Data required	**Rapid Assessment**	**Intermediate Assessment**	**Comprehensive Assessment**
The maximum concentration at the boundary of the riparian zone is calculated and compared to the South African aquatic ecosystem water quality guidelines. The data required for the calculations are: No information is needed for the rapid assessment. For the intermediate assessment the following is needed: • Initial concentration at the source (C_0). • Length between riparian zone and source (L). This is also used to calculate the dispersivity value (α) • Duration of contamination (t). • Groundwater gradient (dh/dl), effective porosity (n_e) and hydraulic conductivity (K) to determine velocity (v). • Dispersivity (α) and velocity (v) to determine the dispersion coefficient (D). For the comprehensive assessment the following is needed: • Initial concentration at the source. • Positions of source and boreholes. • Duration of and injection rate of contaminant. • Area of source and thickness of aquifer. • Groundwater gradient, effective porosity and hydraulic conductivity to determine velocity. • Dispersion coefficients. • Concentration at point in aquifer together with coordinates of point and time at which concentration was measured.	***No concentration values are needed and no calculations are performed. Refer to Table 7.4.***	The equation used to calculate concentrations is in the following format (Fetter, 1999): $C = \frac{C_0}{2}\left[\operatorname{erfc}\left(\frac{L - vt}{2\sqrt{Dt}}\right)\right]$ where $D = \alpha v$ and $v = \frac{K}{n_e}\left(\frac{dh}{dl}\right)$.	Two calculations will be used for the following contamination sources: • Continuous injection • Slug (once-off) injection. For information concerning the Equations used in this section refer to Appendix C (Section C5). It is important to note that dispersion coefficients must be determined by means the equations in Appendix C (Section C5).

Table 7.4: Membership functions/values for decision rules for ecological risk assessment

<table>
<tr><th colspan="4">WATER QUANTITY ASSESSMENT[1]</th></tr>
<tr><td colspan="2">Difference between groundwater level (mbgl) and root depth (m)
The membership function is a cosine graph

</td><td colspan="2">Base flow (BF) versus abstraction (Q)
The membership function is a cosine graph

</td></tr>
<tr><td>Unfavourable limit</td><td>Favourable limit</td><td>Unfavourable limit</td><td>Favourable limit</td></tr>
<tr><td>≤ 0.5</td><td>2</td><td>$Q \geq 0.3$ BF[3]
$Q \geq 0.2$ BF[4]
$Q \geq$ BF – IFR[5]</td><td>Q = 0</td></tr>
<tr><th colspan="2">Type of dependence</th><th colspan="2">Surface water – groundwater interaction</th></tr>
<tr><th>Range</th><th>Membership value</th><th>Range</th><th>Membership value</th></tr>
<tr><td>Obligatory: Entirely
Obligatory: Highly
Obligatory: Proportionally
Facultative</td><td>0.0
0.2
0.4
0.8</td><td>Effluent
Intermittent
Influent
Detached</td><td>0.0
0.2
1.0
1.0</td></tr>
<tr><th colspan="4">Uniqueness of ecosystem</th></tr>
<tr><th colspan="2">Range</th><th colspan="2">Membership</th></tr>
<tr><td colspan="2">Endangered
Sensitive indigenous
Indigenous
Alien</td><td colspan="2">0.0
0.2
0.5
1.0</td></tr>
<tr><th colspan="4">WATER QUALITY ASSESSMENT</th></tr>
<tr><th colspan="4">Toxins[2]</th></tr>
<tr><td>Unfavourable limit
Acute effect value (AEV)</td><td>Favourable limit
0</td><td colspan="2">
</td></tr>
<tr><th colspan="4">Duration of contamination</th></tr>
<tr><th colspan="3">Range</th><th>Membership value</th></tr>
<tr><td colspan="3">Contamination may be seconds, minutes or hours</td><td>0.9</td></tr>
<tr><td colspan="3">Contamination occurs at intermittent periods < 2 years</td><td>0.6</td></tr>
<tr><td colspan="3">Contamination > 90 days and < 2 years</td><td>0.6</td></tr>
<tr><td colspan="3">Contamination occurs at intermittent periods >2 years</td><td>0.3</td></tr>
<tr><td colspan="3">Continuous contamination > 2 years</td><td>0.0</td></tr>
</table>

[1]Membership functions are documented in Appendix E, Section E4. [2]For the rapid assessment, concentrations are not needed therefore the contaminants are classified according to death (1.0), effects common (0.75), long-term effects (0.6) and few effects (0.2). [3]For rapid assessment. See Appendix E, Section E5. [4]For intermediate assessment. [5]For comprehensive assessment inflow

stream requirements (IFRs) are needed.

The ecological risks (quantity and quality) are calculated using Equation 3.1. The decision rules on which the calculations are based are listed in Appendix F, Section F5. The overall ecological risk can then be calculated by means of the following Equation:

$$Risk_{ecology} = \max(Risk_{quantity}, Risk_{quality})$$

7.5 EXAMPLE

The Mutshindudi River Catchment, A91G, (Figure 7-3) is located some 50 km east of Louis Trichardt in the Northern Province of South Africa. The study area is underlain mainly by Soutpansberg Group Rocks consisting of the Sibasa Basalt Formation and the Fundudzi Formation. Diabase intrusions are common throughout the study area, occurring as sills and dykes.

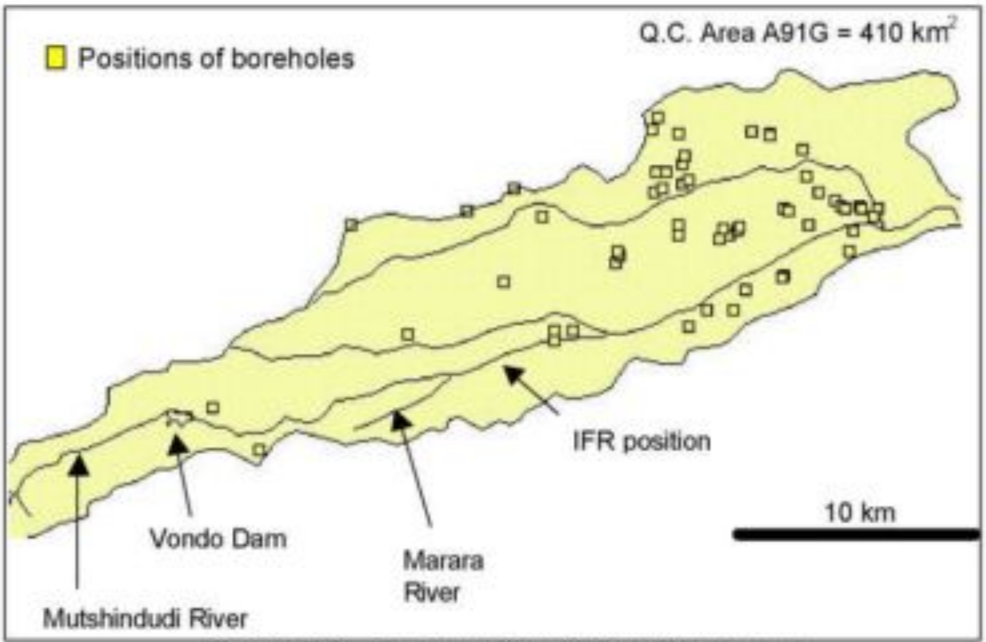


Figure 7-3. The Mutshindudi River Catchment

As the rapid assessment is elementary, and the intermediate and comprehensive assessments are similar, only a comprehensive assessment will be conducted in this example. Where the required data is not available for the comprehensive assessment, hypothetical values are assumed.

The assessment is divided into a quantity and quality assessment. The data required for each are listed in Table 7.5.

Table 7.5: Data required for ecological risk assessment

Input	Value
Quantity Assessment	
Base flow estimation (m^3/s)	0.4170
Inflow stream requirements for the same period as the monthly flow values (m^3/s)	0.275
Perennial/non-perennial	Perennial
Vegetation type to determine average root depth	Trees - overall
Water table (mbgl)	8
Groundwater-surface water interaction	Effluent
Plant dependence on groundwater	Obligatory entirely
Uniqueness of ecosystem	Indigenous
Abstraction rates	BH1 = 5 L/s, BH2 = 5 L/s
Quality Assessment	
Duration of pollution	Polluting > 90 days and < 2 years
Contaminant	Ammonia
Position of source	X = 0, Y = 0
Initial concentration of contaminant (mg/L)	10
Distance between source and riparian zone (m)	500
Area of source (m^2)	1
Dispersivity[1] (m)	70
Hydraulic conductivity (m/d)	6
Porosity	0.1
Groundwater gradient	0.02
Evaluation time (days)	360

[1]Or the user can enter a borehole with coordinates, a concentration value and the time at which the concentration was measured. The DT will then calculate the dispersivity value.

The input screen and results of the quantity risk assessment can be seen in Figure 7-4.

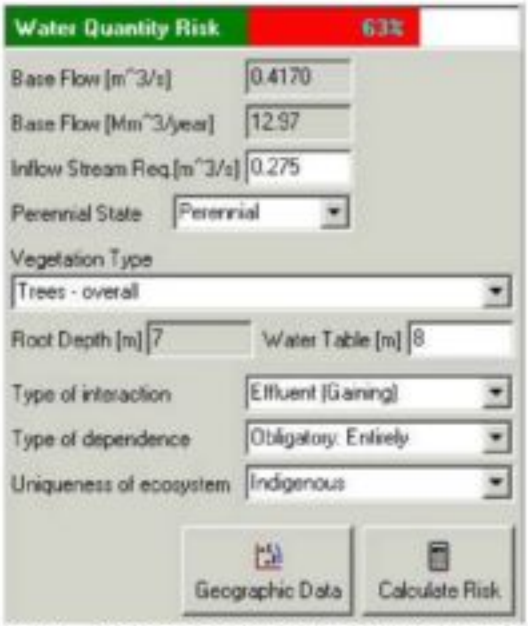


Figure 7-4. Comprehensive quantity risk assessment

The input screen and results of the quality risk assessment are shown in Figure 7-5.

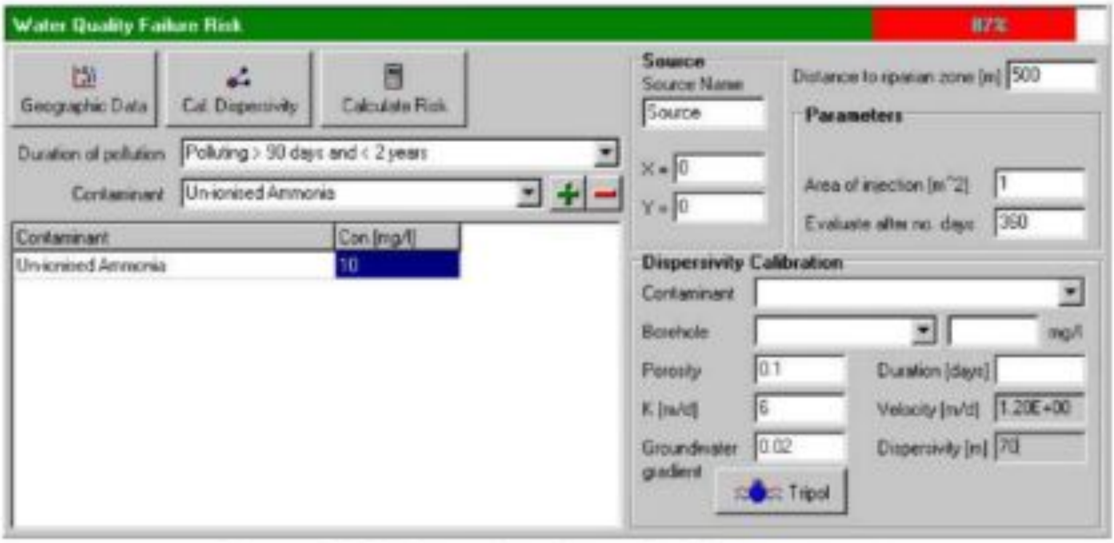


Figure 7-5. Comprehensive quality risk assessment

The total ecological risk can then be calculated as:

$$\text{Risk}_{\text{ecology}} = \max(\text{Risk}_{\text{quantity}}, \text{Risk}_{\text{quality}}) = 87\%$$

This risk indicates that there is an 87% chance that an aquatic ecosystem is going to be impacted by decreases in groundwater quantities and/or groundwater contamination.

An important note: The option polluting > 90 days and < 2 years is treated as a slug source, so once the contaminant has moved passed a certain point in the aquifer the risk will decrease. For example if the distance to the riparian zone is decreased to 10m, then the risk will decrease to 26% as the center of the contaminant has passed that point after a year. However if the time is reduced to 30 days, the risk will increase to 99%, 10m from the source.

CHAPTER 8

Protection

8.1 INTRODUCTION

Since the early 1980's geohydrologists and engineers have developed a number of techniques for protecting groundwater. The National Water Act (Act No 36, 1998) defines protection in relation to a water resource as:

a) *Maintenance of the quality of the water resource to the extent that the water resource may be used in an ecologically sustainable way;*
b) *Prevention of the degradation of the water resource; and*
c) *The rehabilitation of the water resource.*

It is clear that protection refers to both the quantity and quality of a resource. The government places so much emphasis on protection of water resources that a whole chapter of the National Water Act is dedicated to discussing this topic. In this chapter, it is also clearly stated that protection is divided into two categories: measures to prevent the pollution of water resources, and measures to remedy the effects of pollution on water resources. These two categories will be discussed in more detail in the following sections.

8.2 PROTECTION

According to the National Water Act (Act No 36, 1998), the persons who own, control, occupy or use the land in question are responsible for taking measures to prevent pollution of water resources, and in the context of this report, groundwater resources. If these measures are not taken, the catchment management agency concerned may itself do whatever is necessary to prevent the pollution and to recover all reasonable costs from the responsible persons.

There are two major approaches to prevention or minimisation of contamination, namely:

- source control measures and
- groundwater or borehole protection measures.

Each of these will be discussed in more detail in the following sections.

8.2.1 Source Control Measures

The objective of source control is to reduce or eliminate the volume of contamination, thereby eliminating or minimising groundwater pollution. Source control measures include the physical removal or reduction of the source of contamination or the containment of the source. If the source is removed, contamination can no longer migrate from it. It is however important to note that the excavation and removal of hazardous waste materials must be done in a manner that protects the health and safety of the workers and the public. The risk and costs of moving material must be weighed up against the risk and costs of remediation involved to leaving it in place. If the source of contamination cannot be economically or technically removed, then it may be possible to contain it. A groundwater cutoff wall can divert groundwater flow from passing through a contaminant source. If there is no recharge or flow through the cutoff wall, then the water table within the wall will remain flat (see Figure 8-1). However, there is generally leakage through the cover or cutoff walls, so some abstraction boreholes will be needed within the wall to prevent build-up of water within the walls. If the cutoff walls are extended far enough around the source and contaminant plume, then remediation may proceed without concern that it will spread further. In most cases it will also be necessary to construct a cover over the contamination to prevent the infiltration of precipitation. Table 8.1 is a summary of some of the cutoff walls that can be used to contain pollution.

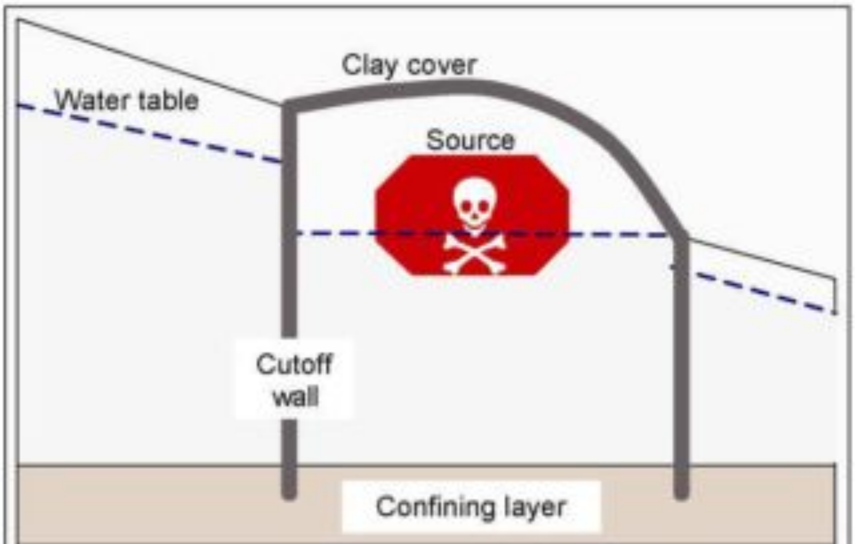


Figure 8-1. Side view of a cutoff wall surrounding contamination source and plume.

Table 8.1: Types of cutoff walls *(Summarised from http://www.clu-in.org/remed1.cfm)*

Type of cutoff wall	Description
Bentonite slurry	These subsurface barriers consist of a vertically excavated trench that is

	filled with a slurry. The slurry hydraulically shores the trench to prevent collapse and forms a filter cake to reduce groundwater flow. Slurry walls are often used where the waste mass is too large for treatment and where soluble and mobile constituents pose an imminent threat to a source of drinking water. Most slurry walls are constructed of a soil, bentonite, and water mixture; walls of this composition provide a barrier with low permeability and chemical resistance at low cost. Other wall compositions, such as sheet piling, cement, bentonite, and water, may be used if greater structural strength is required or if chemical incompatibilities between bentonite and site contaminants exist. Slurry walls are typically placed at depths less than 15 m and are generally 0.6 to 1.2 m thick. Soil-bentonite backfills are not able to withstand attacks by strong acids, bases, salt solutions, and some organic chemicals.
Cement-based grout	The subsurface barrier technology is a combination of techniques to install and verify the integrity of a barrier. The grouts must have the proper hardening time considering the method of injection. This will ensure the grout does not harden too quickly so that it reaches the areas where it is needed and it does not harden too slowly that it spreads out too thinly. Barriers are limited by the depth and directional control of the drilling technology and limited by the inability of nonintrusive techniques to verify barrier continuity.
Sheet piling	A sheet piling barrier can be made from a variety of materials: wood, recast concrete, and steel. Steel is the most common material because of its high durability, low cost, and high flexibility. Sheet pilings are constructed by driving individual sections of interlocking steel sheets into the ground with impact or vibratory hammers to form an impermeable barrier. The retaining sheet pile walls flex from water or lateral earth pressure applied to them. The flexure tightens the interlocks making the connection more water resistant. The process is not suitable for stiff clay or soils containing cobbles and boulders.
Synthetic membranes	Synthetic membranes used for vertical cutoff walls are generally made from high-density polyethylene; however, other polymers have been used. Membrane sheets can be continuous, but usually finite length panels that interlock are preferred.

8.2.2 Protection of a groundwater resource

Groundwater for basic human needs and aquatic ecosystems need to be protected, the most important being water for basic human needs. This section will therefore focus on the protection of groundwater (quantity and quality) for basic human needs, and aquatic ecosystems.

8.2.2.1 Protection of basic human needs boreholes and springs

The parameter that should be considered when determining a protection area around a borehole is the capture area of the borehole which will be used to estimate its safe yield and to determine the impact of pollution (Van Tonder and Dennis, 2000).

A capture area or zone is defined as the area contributing flow to that particular borehole. If the groundwater heads are flat, the capture zone is radially symmetrical, centered on the borehole and extending as far as the cone of depression. However, if there is a slope in the groundwater heads, there is regional groundwater flow and the capture zone is asymmetrical, with the greatest extent in the up-gradient direction. The shape of the capture zone is a function of the average linear groundwater velocity, the quantity of water being pumped from the aquifer, and the distribution of hydraulic conductivity. The up-gradient extent of the capture zone depends on the length of time over which the pumping occurs (Fetter, 1999). Traditionally numerical models have been used to determine capture zones. However there may not always be sufficient data to use this methodology. Therefore if there is insufficient data available, wellhead protection areas (WHPAs) need to be delineated. A WHPA can be defined as the surface and subsurface area surrounding a borehole or wellfield, supplying basic human needs, through which contaminants are reasonably likely to move and reach such a borehole or well field. In many cases it is difficult to protect the whole area, therefore various zones are established within the area. These zones are defined and discussed in Table 8.2.

Table 8.2: Zones within WHPA *(Summarised from Braune, 2000; EPA, 2001 and Boulding, 1995)*

Zone	Definition	Constraints	Calculation
Zone 1: Accident prevention or sanitary protection.	Highly protected area around the borehole or spring. Its purpose is to protect the borehole or spring from the direct introduction of contaminants into the borehole and its immediate area from spills, surface runoff, or leakage from storage facilities or containers. Potential contaminant sources in Zone 1 should be strictly monitored.	Vehicle and pedestrian traffic, Agriculture, All constraints of zone 2 & 3.	Determine 50 day travel time: $r_{50} = \frac{50K\frac{dh}{dl}}{n_e}$ where K = hydraulic conductivity, dh/dl = groundwater gradient and n_e = effective porosity[1]. In the case of a fractured rock system, the porosity must reflect the nature of the system. This can be determined from tracer tests discussed in Chapter 2, Section 2.3.3.
Zone 2: Attenuation	Is established to protect a borehole from contact with pathogenic micro-organisms (e.g. bacteria and viruses) which can emanate from a source (eg septic system) located close to the borehole, as well as to provide emergency response time to begin active cleanup and/or implementation of contingency plans should a chemical contaminant be introduced into the aquifer near the borehole.	Workshops, Farm stables and sheds, Stockyards of building material, Roads and railways, Parking lots, Car washes, Cemeteries, Mining, Fuel storage, Small informal settlements with pit latrines, Junk yards, All constraints of zone 3.	2 year TOT[2] radius area. The radius is calculated as: $r_{TOT} = SF\sqrt{\frac{Qt}{n_e D\pi}}$ where Q = Annual average pumping rate, n_e = effective porosity[1], D = saturated thickness of aquifer, t = 2 years time of travel and SF = safety factor (=1.3 when all values are known, = 1.5 when there are some unknowns). In the case of a fractured rock system, the porosity must reflect the nature of the system. This can be determined from tracer tests discussed in Chapter 2, Section 2.3.3.

Table 8.2 continued

Zone 3: Remedial action	Is designed to protect the borehole from chemical contaminants that may migrate to the borehole; it typically includes a major portion of the recharge area or the capture zone.	Mass livestock, Wastewater and sewage treatment, Hospitals, Airports and military facilities, Trucking and bus facilities, Waste sites, Oil refineries, Chemical plants and nuclear reactors, Deposition and underground storage of water-endangering substances, Pipelines for water-endangering substances, Large informal settlements using pit latrines, Dry cleaning establishments.	5 year TOT radius. The radius is calculated as: $r_{TOT} = SF\sqrt{\frac{Qt}{n_e D\pi}}$ where Q = Annual average pumping rate, n_e = effective porosity[1] D = saturated thickness of the aquifer, t = 5 years time of travel and SF = safety factor (=1.3 when all values are known, = 1.5 when there are some unknowns). In the case of a spring determine radius of influence with t = memory time of the spring. In the case of a fractured rock system, the porosity must reflect the nature of the system. This can be determined from tracer tests discussed in Chapter 2, Section 2.3.3.

[1]If effective porosity is not known the user can use values stored in the DT database. Refer to Appendix C, Section C8.

[2]The TOT criterion bases WHPA delineations on the amount of time it takes groundwater to travel from a point source to a borehole.

8.2.2.2 Protection of groundwater for aquatic ecosystems

When protecting aquatic ecosystems the user has to consider both the groundwater gradient towards the aquatic ecosystem and the quality of this water. The protection is therefore divided into protecting quantity and quality.

Protection of groundwater flow towards an aquatic ecosystem

The protection of groundwater flow towards an aquatic ecosystem is based on the assumption that the groundwater gradient in and around the riparian zone must be able to maintain the requirements of the system. These requirements are known as inflow stream requirements (IFRs) and are set by a team of specialists. If however the IFRs are not known the user can assume that 20% of base flow is necessary to maintain aquatic ecosystems (this value has been obtained from numerous field investigations). In addition the groundwater flow toward an ecosystem will vary from month to month and these variations are usually necessary to maintain the optimal functioning of the ecosystems. It is suggested that the user base the calculations of the groundwater gradient required to maintain the aquatic ecosystem on a high flow and a low flow determination. It would be more accurate to calculate the gradient for every month of the year. The methodologies included in the DT are listed in Table 8.3.

Protection of groundwater quality flowing towards an aquatic ecosystem

There are two methods to protect the groundwater quality flowing toward an aquatic ecosystem, the first being a protection zone around the aquatic ecosystem and the second being a number of constraints on system variables, non-toxic inorganic constituents and nutrients as defined in Chapter 7, Section 7.1. The protection zones and constraints are discussed in more detail in Table 8.3.

Points to consider when protecting groundwater for aquatic ecosystems:

- The protection areas delineated in this section are for ideal conditions, where basic human needs are not an issue. However, when these and other factors become important, the size of the protection area can be changed.
- The surface water body is assumed to be dependent on groundwater.

Table 8.3: Protection of aquatic ecosystems

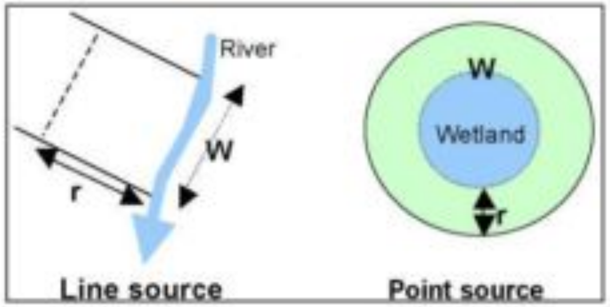
QUANTITY OF GROUNDWATER FLOWING TOWARD THE AQUATIC ECOSYSTEM	
Data required	**Method**
• Groundwater component of base flow (BF) or IFR. • Transmissivity (T). • Length of surface water body under investigation (W).	The groundwater gradient that must be maintained is: $i = \frac{QB \times 0.8}{TW}$ or if the IFR is known $i = \frac{IFR}{TW}$ In the case of a fractured rock system, the transmissivity must reflect the nature of the system.
QUALITY OF GROUNDWATER FLOWING TOWARD THE AQUATIC ECOSYSTEM	
Data required	Method
Protection area/zone • Groundwater component of base flow (BF). • Effective porosity (n_e). • Saturated aquifer thickness (D). • Safety factor (SF) (=1.3 when all values are known, = 1.5 when there are some unknowns).	The length r of the protection area is calculated as: $r = SF\sqrt{\frac{2BF}{n_e D\pi}}$ In the case of a fractured rock system, the porosity must reflect the nature of the system. This can be determined from tracer tests discussed in Chapter 2, Section 2.3.3. 

Table 8.3 continued

Constraints taken from South African Water Quality Guidelines (DWAF, 1996).	*1. System variables* Dissolved oxygen: Must not drop below the arithmetic mean of the daily minimum instantaneous concentrations measured at hourly intervals over 7 consecutive days AND the lowest instantaneous concentration recorded in a 24-hour cycle, or the instantaneous concentration at sunrise. Target value 80 – 120% of saturation concentration. pH: The pH values should not be allowed to vary from the range of the background pH values for a specific site and time of day, by > 0.5 of a pH unit OR by > 5%, and should be assessed by whichever estimate is more conservative. Temperature: Water temperature should not be allowed to vary from the background average daily water temperature considered to be normal for that specific site and time of day, by > 2°C OR by > 10%, whichever estimate is more conservative. **2. Non-toxic constituents** Total dissolved solids (TDS): TDS concentrations should not be changed by > 15% from the normal cycles of the water body under unimpacted conditions at any time of the year AND the amplitude and frequency of natural cycles in TDS concentrations should not be changed. Total suspended solids (TSS): Any increase in TSS concentrations must be to < 10% of the background TSS concentrations at a specific site and time. **3. Nutrients** Nitrogen: The inorganic nitrogen concentrations should not be changed by more than 15% of unimpacted conditions at any time of the year AND the trophic status of the surface water body should not increase above its present level AND the amplitude and frequency of natural cycles in inorganic nitrogen concentrations should not be changed. Phosphorus: The inorganic phosphorus concentrations should not be changed by > 15% from unimpacted conditions at any time of the year AND the trophic status of the water body should not increase above its present level AND the amplitude and frequency of natural cycles in inorganic phosphorus concentrations should not be changed. **4. Toxic constituents** The concentrations of none of the toxic nutrients may exceed the chronic effect value as documented in the Water Quality Guidelines for Aquatic Ecosystems.

8.2.3 Examples

When selecting the remediation and protection screen appears (Figure 8-2). By selecting "Borehole Protection", the screen appears where the values can be entered to calculate the various protection zones as discussed in Table 8.2. The definition of each zone together with the constraints appears on the left-hand side of the screen. When selecting "Geographic Data" the boreholes under consideration can be entered. The DT will then tell the user in which assessments the required data has been used under the "mapping" boxes. The user can then decide which assessment's data to use for these calculations. The user can also enter the data directly in the table provided.

Take for example borehole UO5 situated at the Campus Site of the University of the Free State. The data have directly been entered into the table (Figure 8-3). The radius of the protection zones then appears after "Calculate Zones" has been selected. By selecting "Geographic Data" the user can view the zones (Figure 8-4).

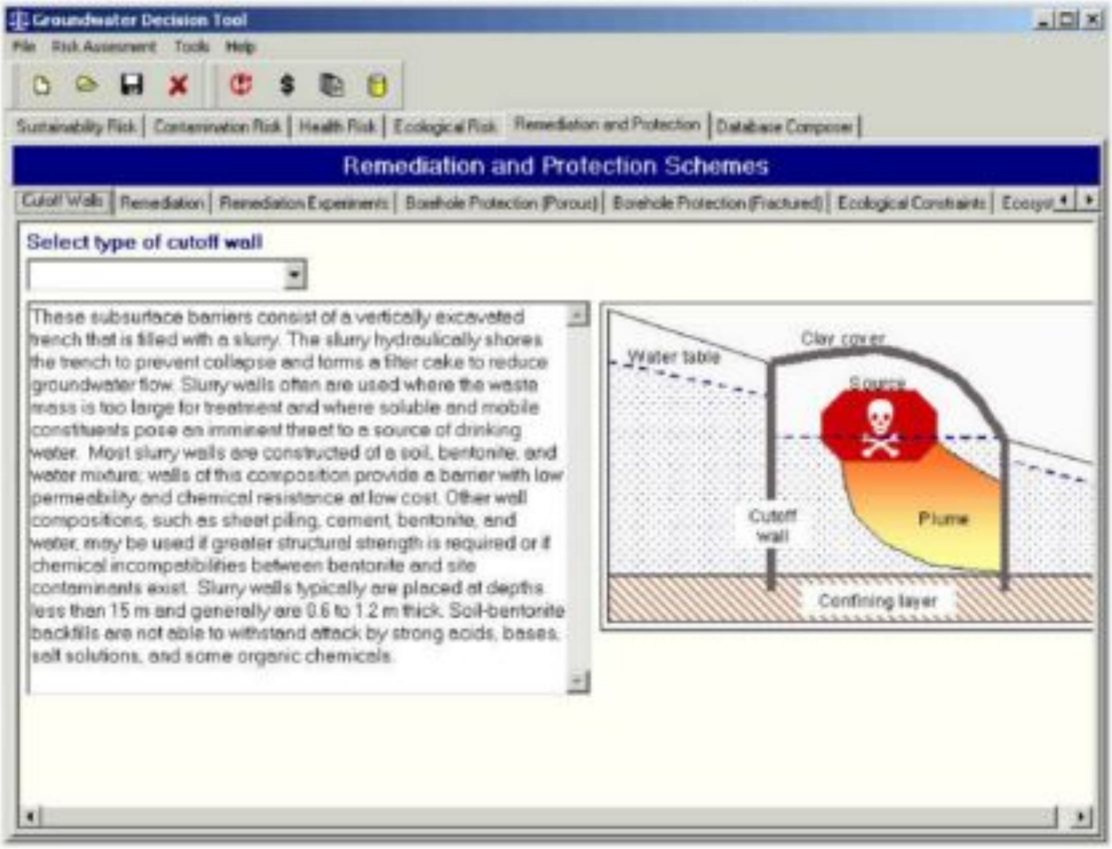


Figure 8-2. Initial remediation and protection screen

Figure 8-3. Protection zone calculations for borehole UO5 when pumping at 1 L/s

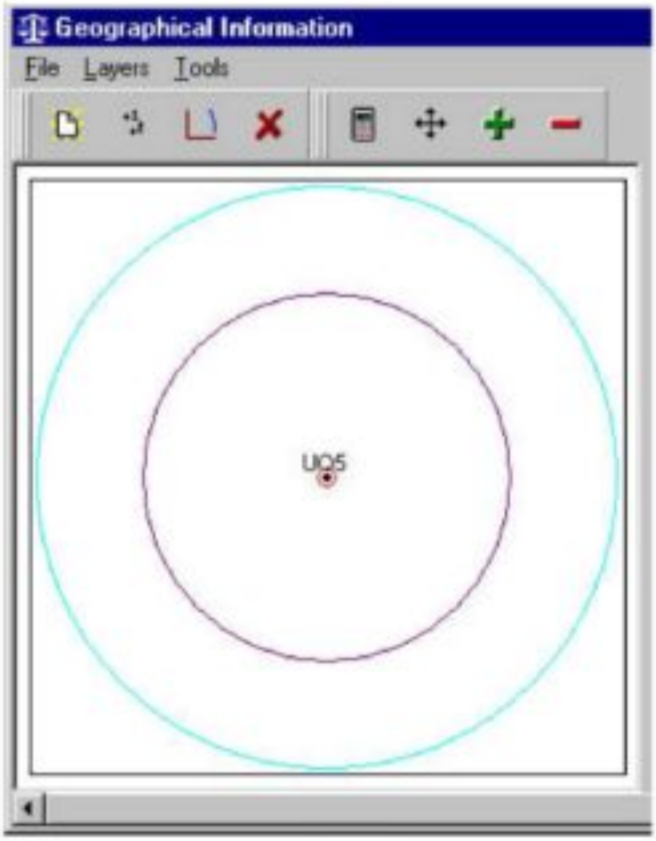


Figure 8-4. Graphical representation of the protection zones for borehole UO5

It must be noted that the protection zones shown in Figure 8-4 are based on a matrix transmissivity and porosity value. However if the transmissivity of the fracture zone is taken as 100 m^2/d, the porosity as 49% and the saturated thickness as 2 m, the following values are obtained for the different zones of protection:

- Zone 1 = 154.29 m
- Zone 2 = 213.22 m
- Zone 3 = 337.13 m

By selecting "Ecosystem Protection" the groundwater gradient towards an aquatic ecosystem and zone of protection can be determined. For example take the values used in the Mutshindudi River Catchment assessments. The results of the protection options are shown in Figures 8-5 and 8-6.

Figure 8-5. Groundwater gradient calculation

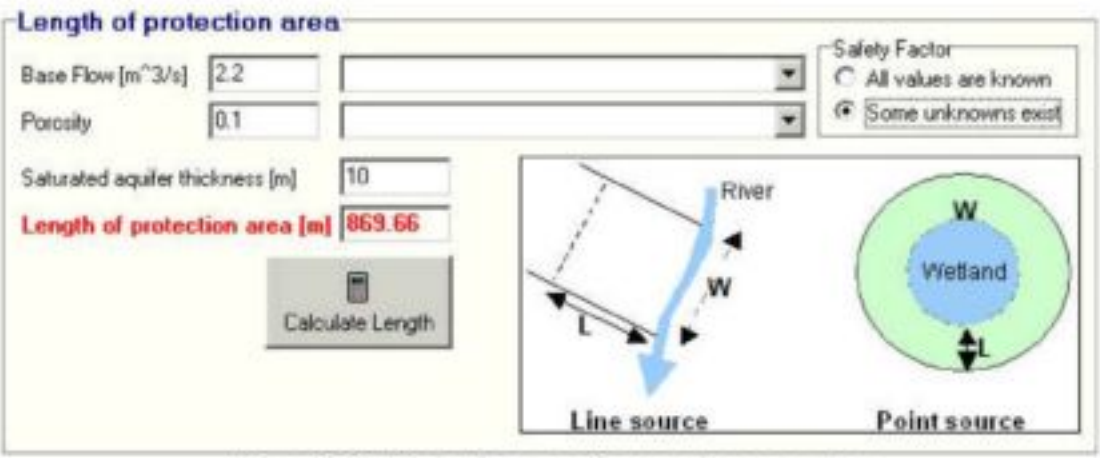


Figure 8-6. Protection zone for aquatic ecosystem

The user must realise that the gradient and protection zone must be applied in conjunction with the constraints, viewed when selecting "Ecological Constraints".

8.3 REMEDIATION

Remediation refers to the cleanup or other methods used to remove or contain a toxic spill or hazardous materials from a contaminated site. Remediation is a complex subject that is site-specific and will therefore not be discussed in detail. The section on remediation will be divided into two. The first section will focus on some experiments conducted concerning remediation. These experiments will give the reader an indication of the possible success of a remediation project. The second section will discuss various remediation options.

8.3.1 Possibilities of remediation

Experiments were conducted to determine the amount of NaCl and Na_2SO_4 that can be retrieved from various sandstones, shales and a quartzite. The various cores were soaked in strong NaCl and Na_2SO_4 solutions. The concentrations of NaCl and Na_2SO_4 used are 300000 mg/L and 2500000 mg/L deionized water respectively. The core was then placed in deionized water and the increase in electrical conductivity was measured with time. The results of can be seen in Figure 8-7.

The results of the experiments are listed in Table 8.4. It is clear that the success of remediation can vary from about 50% to almost 100%. Most of the experiments were performed more than once to ensure the validity of results. The exceptions were sandstone with a porosity of 4.4% and shale with a porosity of 0.83%, due to the fact that limited core available. It is important to note that factors such as continuous flushing with deionized water and natural attenuation can improve the success rate.

Table 8.4: Results of remediation experiments

Formation	Porosity (%)	Amount NaCl retrieved (%)	Amount Na_2SO_4 retrieved (%)
Sandstone (coarse grain)	10.9	-	91
Sandstone (medium grain)	7.1	95	70
Sandstone (medium grain)	6.1	96	75.5
Sandstone (fine grain)	4.4	85	77
Shale	1.12	-	58
Shale	0.92	93.5	63
Shale	0.83	92	49
Quartzite	0.19	96	65

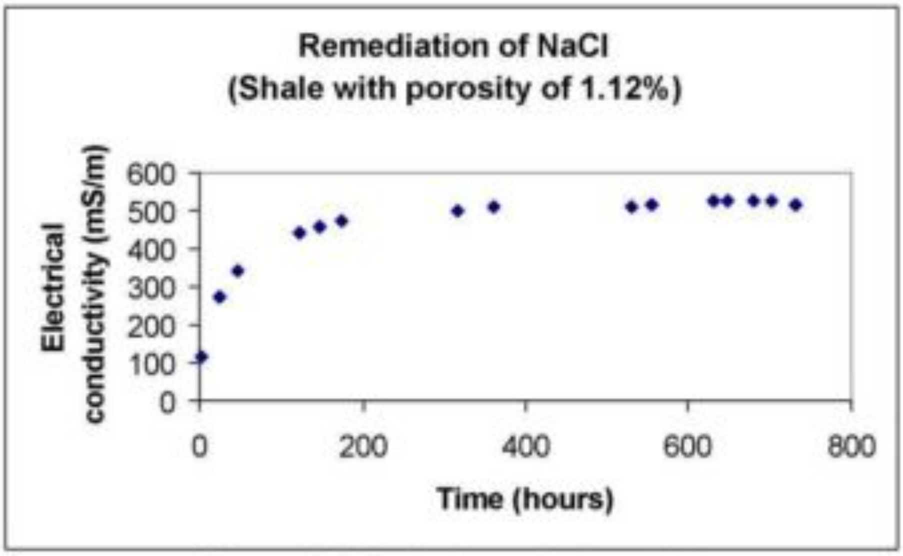


Figure 8-7. Typical remediation graph

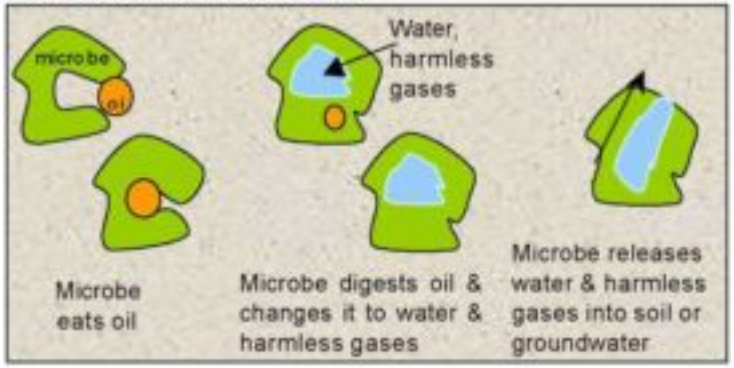
The information from remediation experiments will be stored in the DT where the user will be able to browse the information.

8.3.2 Remediation options

Remediation options are site-specific and therefore are not discussed in detail in this report. However the DT does contain information on most of the remediation techniques available (refer to Table 8.5). The user will be able to browse this data.

Table 8.5: Remediation options *(Summarised from http://www.clu-in.org/remed1.cfm)*

Remediation option	How it works	More information

Bioremediation	Natural clean up of harmful chemicals in groundwater. Microbes present in the soil and groundwater digest certain chemicals (eg petrol and oil) and change them into water and harmless gases.
	
	
		• Takes advantage of natural processes.
		• Do not have to excavate or pump.
		• Prevents the release of harmful gases into the air.
		• Does not require much equipment or labor.
		• The time it takes depends on:
		  ○ Type and amount of chemicals present.
		  ○ Size and depth of polluted area.
		  ○ Type of soil and the conditions present.
		On average it can take a few months to several years.

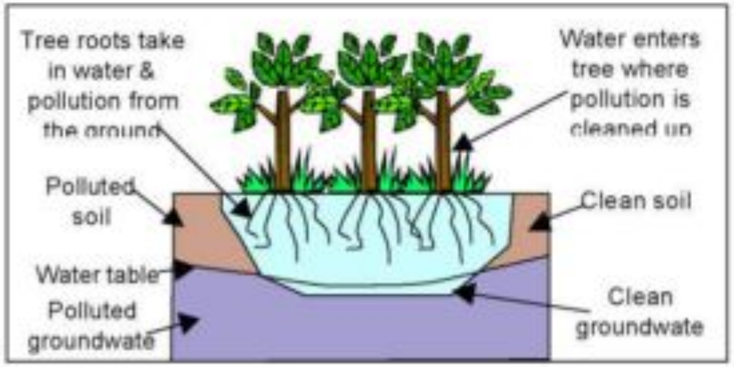
Phytoremediation	Uses plants to clean up pollution (metals, pesticides, explosives & oil). Plants also prevent wind, rain and groundwater from carrying pollution away from site. Plants remove harmful chemicals when their roots take in water, therefore they can clean up chemicals as deep as their roots grow. Only effective in root zone.  	Once inside plant chemicals are: • Stored in roots, stems or leaves. • Changed into less harmful chemicals. • Changed into gases that are released when the plant transpires. The method can be harmful to insects, animals and humans eating plants or release harmful gases into the atmosphere. The advantages are: • Does not require much equipment or labor. • Trees and plants make site attractive. • Do not have to excavate or pump. The time it takes depends on: • Type and number of plants zused. • Type and amount of chemicals present. • Size and depth of polluted area. • Type of soil and the conditions present. On average it takes many years to clean up a site.

Table 8.5 continued

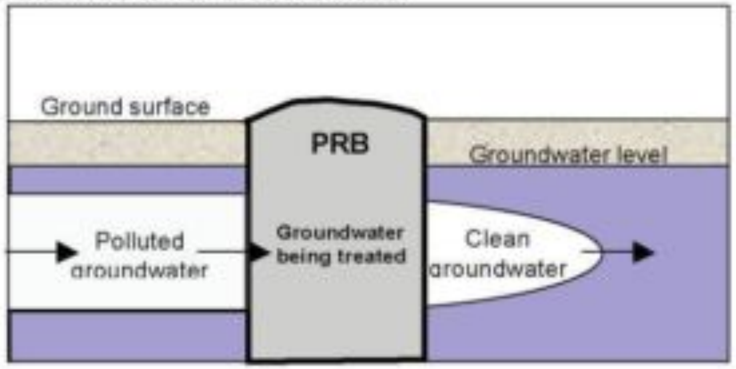
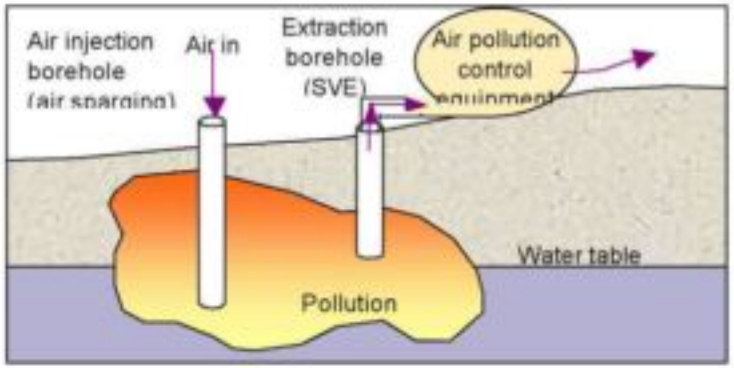
Permeable reactive barriers (PRB)	A PRB is a wall built below the surface to clean up groundwater. The wall allows groundwater to flow through while reactive material in the wall traps and changes harmful chemicals to harmless chemicals. 	Advantages: • No moving parts, no noise. • Polluted water cleaned up underground. • No need to pump. • No equipment above ground so site can be used. Disadvantage: • Some polluted soil must be removed to build barrier. Works best at sites with loose, sandy soil and a steady flow of groundwater. The pollution must not be deeper than 15 m. The time it takes depends on the type and amount of pollution and the rate at which groundwater moves.
Soil vapor extraction (SVE) and air sparging	SVE removes harmful chemicals in the form of vapors from the soil above the water table by means of a vacuum. Air sparging uses air to help remove harmful vapors from polluted soil and groundwater below the water table. Both work best on solvents and fuel. 	Advantages: • Faster than natural processes. • Boreholes and equipment are easy to maintain. • Reaches greater depths than methods involving digging up soil. • Effective in removing any type of pollution that can evaporate. • Helps clean up pollution by encouraging the growth of microbes. Disadvantages: • Must ensure harmful vapors are collected and disposed of properly. • Requires drilling of extraction and air injection boreholes in the polluted area. Works best in loose soils – like sand and gravel. The time it takes depends on the size and depth of polluted area and, the type of soil and the conditions present. The injected air can be heated to speed up the process. Heated soil helps evaporate chemicals faster. Other sources of heat (eg steam or hot water) can be pumped into injection boreholes to heat up the soil.

Table 8.5 continued

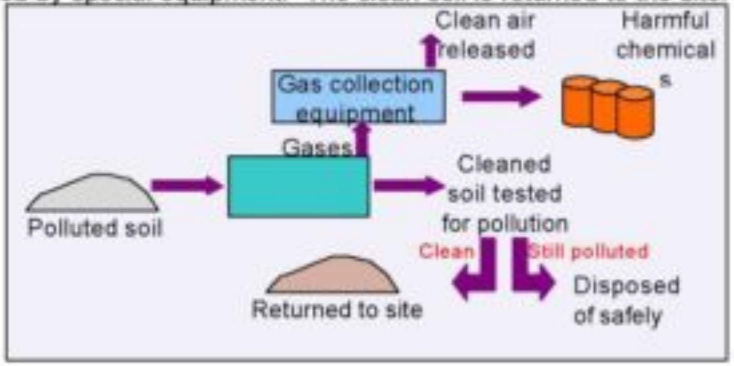
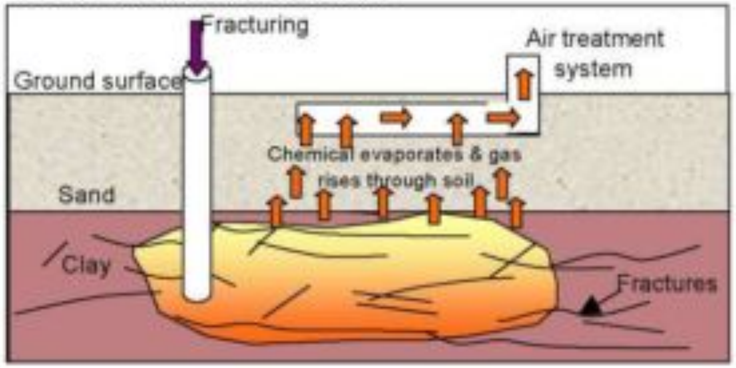
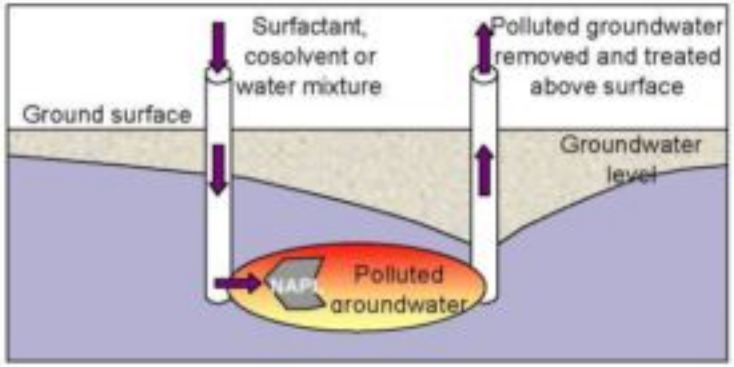
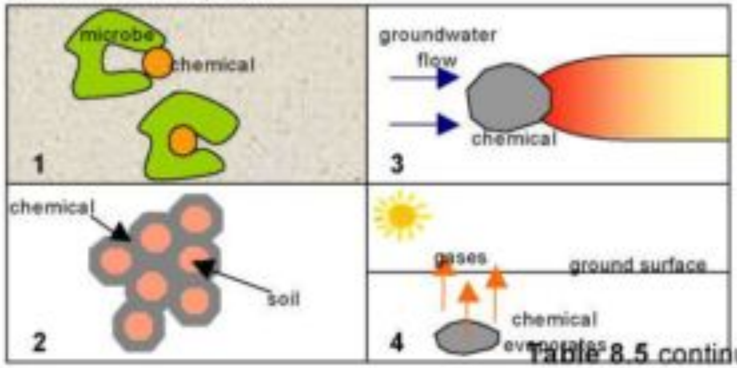
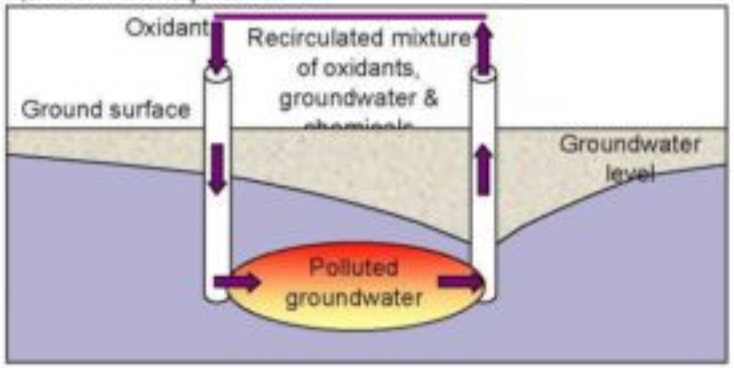
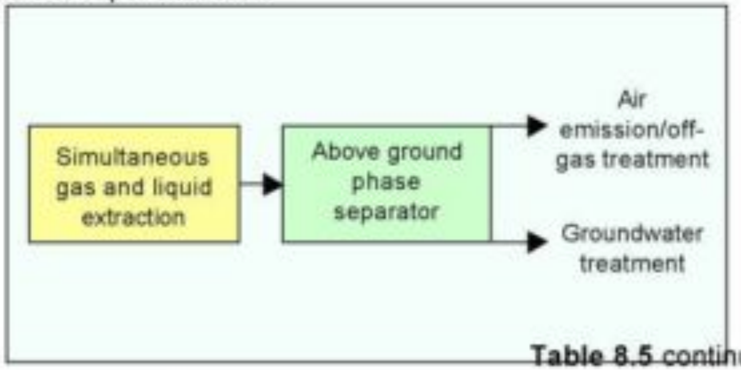
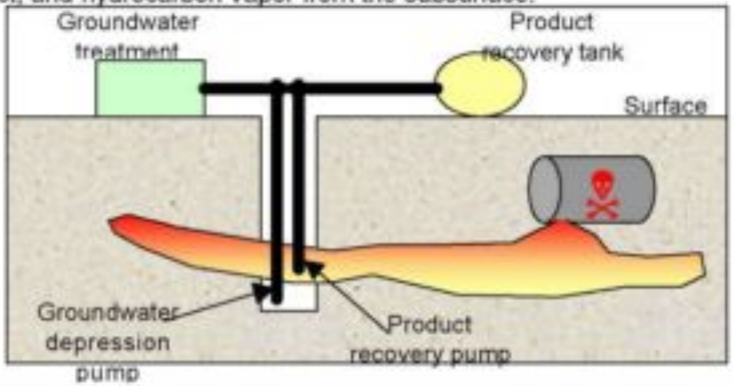
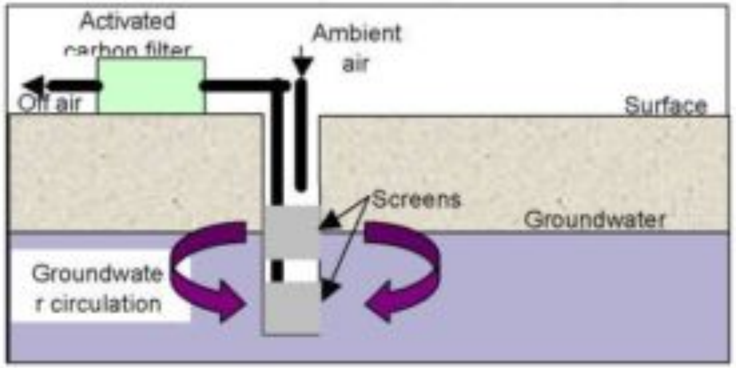
Thermal desorption	This method removes harmful chemicals from soil and other material like slug and sediment by using heat to change chemicals into gases. These gases are collected by special equipment. The clean soil is returned to the site. 	Thermal desorption works well at sites with dry soil and certain types of pollution such as fuel oil, coal tar, chemicals that preserve wood and solvents. Advantages: • Faster clean up method than most. • Equipment often costs less to build and operate than equipment for other clean up methods using heat. Disadvantage: • Soil must be transported off-site, which costs money. The time it takes depends on the amount of polluted soil, the condition of the soil and, the type and amount of harmful chemicals present. A system can clean over 20 tons of polluted soil per hour.
Fracturing	Fracturing is used to crack rock or dense soil. It is not necessarily a cleanup method itself, it rather aids other cleanup methods to be more effective. Fractures create paths through which pollutants can travel. These pollutants can then be evaporated out of the soil or the fractures can be intercepted by boreholes and the pollution pumped out. 	Do not conduct near underground pipelines or above-ground structures. Fracturing offers a way of reaching pollution deep in the ground where it would be difficult or costly to dig down so far. Fracturing can reduce the number of boreholes needed for certain cleanup methods, which can save time and reduce cleanup costs. Often fracturing is used to help clean up non-aqueous phase liquids (NAPLs) – chemicals that don't dissolve readily in groundwater. The time it takes depends on the size and depth of the polluted area, the types and amounts of harmful chemicals present, the type of soil or rock and the cleanup method used. Fracturing rock and soil does not take very long – it may only take a few days. However, the actual cleanup may take months or years.

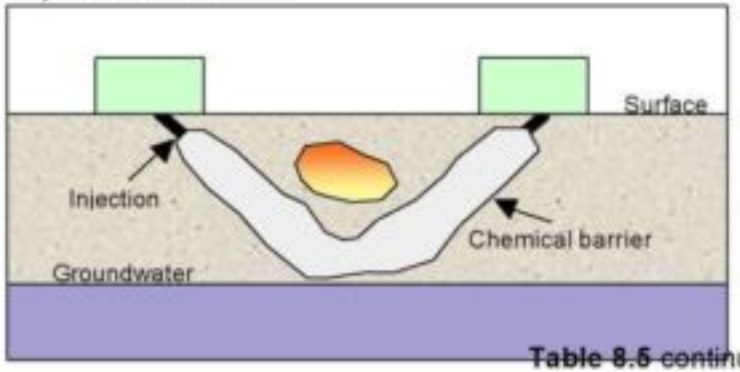
Table 8.5 continued

In situ flushing (steam, water or chemicals)	In situ flushing is a way to clean up harmful chemicals in polluted soil and groundwater by pumping water or chemicals (normally surfactants or cosolvents) into the ground. This helps flush the harmful chemicals from the ground by moving them towards boreholes that pump the chemicals out. Therefore in situ flushing is used to help pump and treat groundwater.  	This method is often used in NAPL remediation. It works best in soil that is very permeable and the soil/rock underneath the polluted area is not very permeable. Advantage: • Avoids the expense of digging up soil for disposal or clean up. Disadvantages: • Workers that handle chemicals pumped down the boreholes must wear protective clothing. • Also surfactant or cosolvent left behind after clean up can be harmful. • Can be expensive and difficult to implement. The target contaminant group for soil flushing is inorganics, including radioactive contaminants. The time it takes depends on the size and depth of the polluted area, the type and amount of NAPL, the type of soil and conditions present and how groundwater flows through the soil/rock matrix. Clean up of a site can take months or years.

Monitored natural attenuation	Relies on natural processes to clean up groundwater or soil. There are four methods: (1) microbes digest chemicals, (2) chemicals sorb to the soil/rock matrix. This does not clean up pollution but keeps it from spreading and (3) some chemicals can evaporate.  Table 8.5 continued	Regular monitoring is needed to make sure pollution does not leave site. Advantages: • No digging or construction and nothing has to be added to clean up pollution. • Less disruptive to neighborhood and environment. • Cleanup workers not in contact with pollution. • Less equipment and labor. Disadvantage: • Monitoring may be costly. The time it takes depends on the size and depth of the polluted area, the type and amounts of chemicals present and the type of soil and conditions present. Clean up usually takes years to decades. These methods are used when other methods do not work.
Chemical oxidation - same methodology as bioventing	This process uses chemicals called oxidants to destroy pollution in soil and groundwater. Oxidants change harmful chemicals into harmless ones like water and carbon dioxide. Chemical oxidation can destroy many types of chemicals like fuels, solvents and pesticides. 	Chemical oxidation can create enough heat to boil water. The heat can cause the chemicals underground to evaporate. Advantages: • Do not have to dig or excavate. • No boreholes are needed. • Saves time and money. • Can reach pollution deep within groundwater system. Disadvantages: • Oxidants are corrosive. • People who work with oxidants must wear special clothing. • Some oxidants can explode. The time it takes depends on the size and depth of the polluted area, how groundwater flows and the type of soil and conditions present. In general chemical oxidation is faster than most methods. Clean up times can be measured in months, rather than years.

Fluid/Vapor Extraction (Two phase extraction)	A vacuum system simultaneously removes liquid and gas from low permeability or heterogeneous formations. It removes contaminants from above and below the water table. The system lowers the water table around the borehole, exposing more of the formation. Contaminants in the newly exposed vadose zone are then accessible to vapor extraction.  Table 8.5 continued	Advantages: • Can remove contaminants more efficiently than pump-and-treat. • Because of the turbulence created during extraction, most of the contaminants in the water are stripped away, and little additional treatment is needed. Disadvantage: • Fluid/vapor extraction requires both water treatment and vapor treatment. The target contaminant groups for fluid/vapor extraction are VOCs and fuels. It is more effective than SVE for heterogeneous clays and fine sands. Fluid/vapor extraction can be combined with bioremediation, air sparging, or bioventing when the target contaminants include long-chained hydrocarbons.
Dual phase extraction	Dual-phase extraction (DPE), also known as multi-phase extraction or vacuum-enhanced extraction, is a technology that uses a high vacuum system to remove various combinations of contaminated groundwater, separate-phase petroleum product, and hydrocarbon vapor from the subsurface.  	The DPE process for undissolved liquid-phase organics, also known as free product recovery, is used primarily in cases where a fuel hydrocarbon lens more than 20 cm thick is floating on the water table. The target contaminant groups for dual phase extraction are VOCs and fuels (eg LNAPLs). Dual phase vacuum extraction is more effective than SVE for heterogeneous clays and fine sands. However, it is not recommended for lower permeability formations due to the potential to leave isolated lenses of undissolved product in the formation.

Bioslurping	Bioslurping combines the two remedial approaches of bioventing and vacuum-enhanced free-product recovery. Bioventing stimulates the aerobic bioremediation of hydrocarbon-contaminated soils. Vacuum-enhanced free-product recovery extracts LNAPLs from the capillary fringe and the water table. **Table 8.5** continued	It is a cost-effective in situ remedial technology. Bioslurping is applicable at sites with a deep ground water table (> 9 m). Disadvantages: • Less effective in low-permeability soils. • Low temperatures slow remediation. • The off-gas requires treatment before discharge. • At some sites, bioslurper systems can extract large volumes of water that may need to be treated. • Since the fuel, water and air are removed, these mixtures may require special oil/water separators or treatment before the process water can be discharged. Operation and maintenance duration for bioslurping varies from a few months to years, depending on site specific conditions.
In-well air stripping	Air is injected into a double-screened borehole, lifting the water in the borehole and forcing it out the upper screen. Simultaneously, additional water is drawn in the lower screen. Once in the borehole, some of the VOCs in the contaminated groundwater are transferred from the dissolved phase to the vapor phase by air bubbles. The contaminated air rises in the borehole to the water surface where vapors are drawn off and treated by a soil vapor extraction system.  	The target contaminant groups for vacuum vapor extraction are halogenated VOCs, SVOCs, and fuels. Variations of the technology may allow for its effectiveness against some nonhalogenated VOCs, SVOCs, pesticides, and inorganics. Typically, in-well air stripping systems are cost-effective. Disadvantages: • Fouling may occur by infiltrating precipitation containing oxidized constituents. • Shallow aquifers limit effectiveness. • Limited to sites with $K > 10^{-2}$ m/d and should not be utilised at sites that have lenses of low-K deposits. • In well air stripping may not be effective at sites with strong natural flow patterns.

Chemical barriers	Chemically based barrier materials involve the use of agents either to reduce permeability of the aquifer or to cause a chemical reaction to detoxify or reduce the mobility of the contaminant.  Table 8.5 continued	Contaminants such as uranium, molybdenum, chromium, arsenic, copper, lead, zinc, and radium can potentially be removed from groundwater. Advantages: • Avoids the water management and groundwater flow interruption problems. • If the permeable barriers are designed to destroy the contaminants rather than absorb them, further management or removal of the hazardous substances becomes unnecessary.
Dams, ditches and drains	Dams are used to control surface water flow that carries either dissolved or suspended contaminants. Ditches are used to control the flow of surface water containing dissolved or suspended contaminants. Less commonly, ditches are used to control groundwater flow. Drains are used to capture groundwater or surface water for the purposes of treatment or containment. Dams, ditches and drains are not a treatment and do not target any specific group of contaminants.	Advantage: • The techniques to construct are well understood. Disadvantages: • The limitations are that in certain soil conditions underflow beneath the dam may occur, and the structure may require constant maintenance. • A limitation of ditches is that in high permeability soils, the water will simply drain into the soil. • The effectiveness of a drain is problematic in complex geohydrological systems.

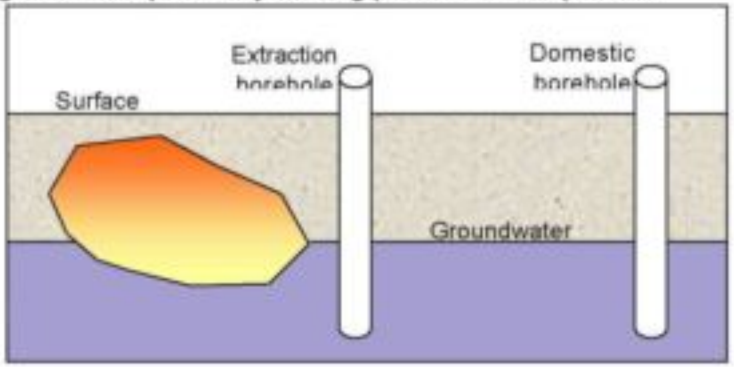
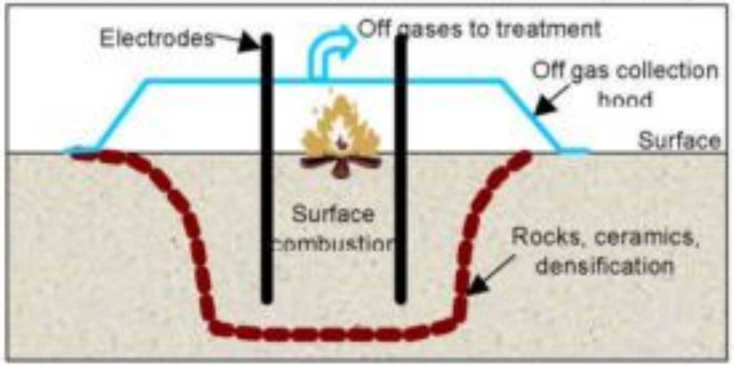
Pumping systems	Extraction boreholes are used to control groundwater flow. The purpose is to contain plume migration by redirecting groundwater from source areas or to control groundwater plumes by creating preferential flow patterns. 	Target contaminant groups are mobile and soluble organics and inorganics. Extraction borehole systems are relatively simple to implement and use standard equipment readily available from multiple sources.
Radioactive decay	Radioactive decay is a natural process where radioactive elements spontaneously emit energetic particles such as electrons or alpha particles. These emissions are harmful. The level of these emissions drops over time as the element reaches a stable state. Incorporating radioactive decay into the remediation strategy also includes monitoring and some form of control.	Radioactive decay is the only method to eliminate the risk of radioactive elements as no treatment exists to eliminate the property of radioactivity.

Table 8.5 continued

In Situ Vitrification	The in situ vitrification (ISV) process can destroy or remove organics and immobilise most inorganics. ISV uses an electric current to melt soil or other earthen materials at extremely high temperatures (1600 to 2000°C) and thereby immobilises most inorganics and destroys organic pollutants by pyrolysis. Process depths up to 6 m have been achieved in relatively homogeneous soils. 	The process can be used on a range of VOCs, SVOCs, and other organics, including dioxins and polychlorinated biphenyls (PCBs), and on most priority pollutant metals and radionuclides. The vitrification product is a chemically stable and leach resistant. The process destroys and/or removes organic materials. Radionuclides and heavy metals are retained within the molten soil. Factors that may limit the applicability and effectiveness of the process include the following: • Subsurface migration of contaminants into clean areas because of soil heating. • Combustible organics.
Conventional excavation	Contaminated material is removed and transported to permitted off-site treatment and/or disposal facilities. Excavation and off-site disposal is a well proven and readily implementable technology.	Disadvantages: • Generation of emissions may be a problem during operations. • Distance from site to the nearest disposal facility will affect cost. • Transportation of the soil through populated areas may affect community acceptability. • Disposal options may be limited.
Pump and treat	Conventional pumping is used for cleanup of organics and inorganics (metals, anions, and radionuclides) in groundwater. The system, consisting of appropriate access boreholes for groundwater extraction, removes contaminants that are dissolved in the water for treatment at the surface. This technology is simple to design and operate, uses standard equipment available from many sources, and treats all types of dissolved contamination.	Advantage: • It can be implemented quickly and is compatible with adjunct technologies. Disadvantages: • Not applicable to fractured rock or clay. • A poor choice for contaminants that adsorb or those with low solubilities.

8.3.3 Example

When selecting  the remediation and protection screen appears (Figure 8-2). Tables 8.1, 8.4 and 8.5 are documented under "Cutoff walls", "Remediation" and "Remediation Experiments". The user can scroll through these tables. For example if the user wants more information concerning bioremediation, the topic has to be selected from the menu in the top left-hand corner of the screen and all the information concerning bioremediation will appear (Figure 8-8).

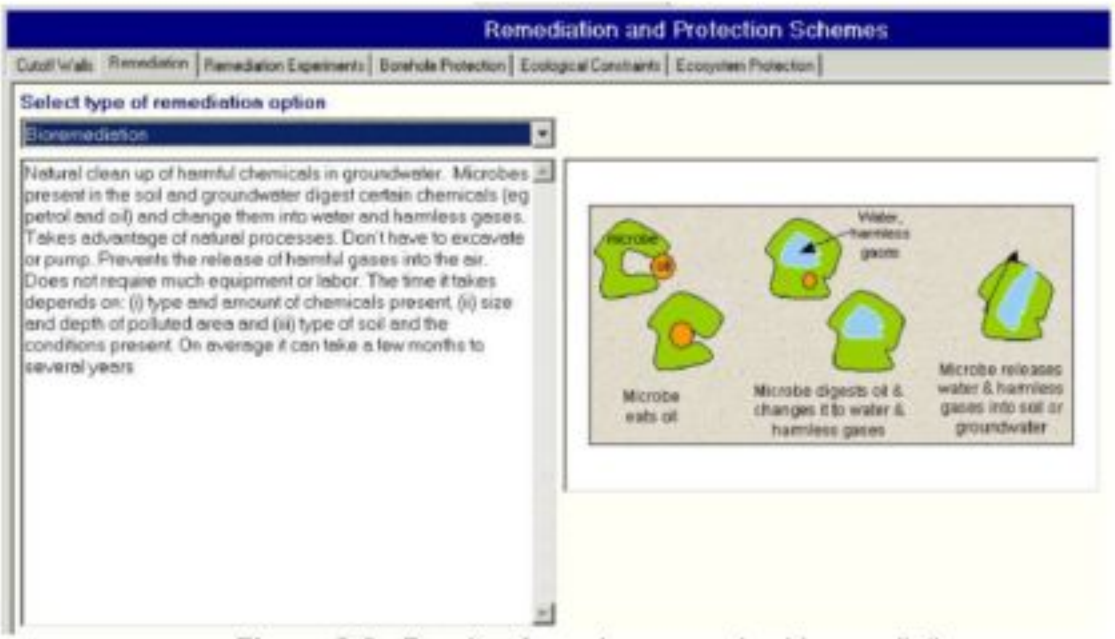


Figure 8-8. Results of enquiry concerning bioremediation

CHAPTER 9

Cost-benefit-risk Analysis

9.1 BACKGROUND

Cost-benefit-risk analysis is widely used as a tool in project appraisal to optimise project design, to assess policies and regulations, and to evaluate decisions entailing more or less measurable economic consequences (Abelson, 1979). Cost-benefit-risk analysis is defined as a set of procedures used for defining, comparing and measuring benefits and costs, which originate from either an investment or an operation. Cost-benefit-risk analysis is flexible and adaptable. However, the assessor should keep the suitability of a cost-benefit-risk analysis for different types of projects in mind before an attempt is made to perform the analysis.

Cost-benefit-risk analysis shapes the framework for decision-making. The fundamental rule for cost-benefit-risk analysis is that decisions are made by decision-makers, and therefore it is an aid for decision-making and not the decision itself.

The advantages of a cost-benefit-risk analysis include transparency, the provision of a framework for consistent data collection and identification of gaps and uncertainty in knowledge. Cost-benefit-risk analysis does not take the "rights" of future generations into account and where environmental protection is desirable, the reasons for the protection are often not quantifiable, for example in the case of social values.

9.2 METHODOLOGY

9.2.1 General

The framework discussed in this section is an early stage monetary risk based cost-benefit-risk analysis, considering both the probability and economical consequences of depleting and contaminating groundwater resources. The framework is aimed at providing a basis for cost-effective decision-making regarding groundwater protection and management actions. A major objective in cost-effective groundwater protection and management should be that of systematically obtaining optimal value from existing geohydrological and other information *before* performing detailed studies (Rosen and LeGrand, 1997). In cost-efficient groundwater protection and management work the costs for protective actions must be in balance with the economical risks of contamination.

9.2.2 Monetary Risk Analysis *(Summarised from Freeze et al., 1990; Janse Van Rensburg, 1992)*

In a decision problem, the benefits, costs and risks of each alternative are taken into account by defining an objective function, ϕ_I, for each alternative i = 1, . . . , n. The objective function should reflect the specific problem and the preferences of the decision-maker, and thus, varies according to the key variables involved (Rosen *et al.*, 1998). The objective function has the general form:

$$\phi_i = \sum_{t=0}^{T} \frac{1}{(1+r)^t} \left[B_i(t) - C_i(t) + V_i(t) - R_i(t)\right] \quad (9.1)$$

where

$B_i(t)$ = Benefits of alternative i in year t

$C_i(t)$ = Costs of alternative i in year t

$V_i(t)$ = Remediation of alternative i in year t

$R_i(t)$ = Risks of alternative i in year t

r = Discount rate

T = Time horizon

The time horizon is relatively short, in the order of 20 – 50 years. The discount rate is the market interest rate on borrowed money.

The objective function presents the net present value of alternative i. The objective of the design must be met to maximise profit or minimise loss.

Remediation can be defined as:

$$V(t) = P_S(t) B_S(t) \gamma(B_S)$$

where

$P_S(t)$ = Probability of success in year t

$B_S(t)$ = Benefits associated with success in year t

$\gamma(B_S)$ = Normalised unity function

The benefits $B_S(t)$ associated with success in a remedial clean up could include permission to reopen operation, removal of legal liabilities and the return of goodwill to the community.

The risk R(t) is defined as the expected costs associated with the probability of failure:

$$R(t) = P_f(t)C_f(t)\gamma(C_f)$$

where

$P_f(t)$ = Probability of failure in year t

$C_f(t)$ = Costs associated with failure in year t

$\gamma(C_f)$ = Normalised unity function

The term C(t) typically represents all fixed and operational costs for each alternative. The $C_f(t)$ term includes the costs that would arise due to the depletion and/or contamination of a groundwater resource. They would include any fines, taxes or charges that might be levied by government for failing to comply with legislation, the costs of litigation, the costs of remedial action; and the value of any revenues foregone should the operation be stopped or curtailed. Goodwill for the community is also included here. When avoiding risks γ is usually greater than one, however for a neutral approach γ is equal to one.

It is important to note that either the failure term OR the success term must be used in Equation 9.1.

9.2.3 Cost-benefit-risk analysis in the decision tool

The decision tool does not calculate risks on a yearly basis, therefore the monetary risk analysis discussed in Section 9.2.2 was simplified and to be included in the decision tool. As there are no calculations concerning the risks involved with the success of remediation in the DT, the option for the success of remediation has not been included.

Time dependency was removed from the calculations, however it is still indirectly taken into account when calculating the benefits and costs. The decision tool takes a neutral approach and assumes:

$$\gamma(C_f) = 1.$$

The final methodology used in the decision tool is summarised in Figure 9-1.

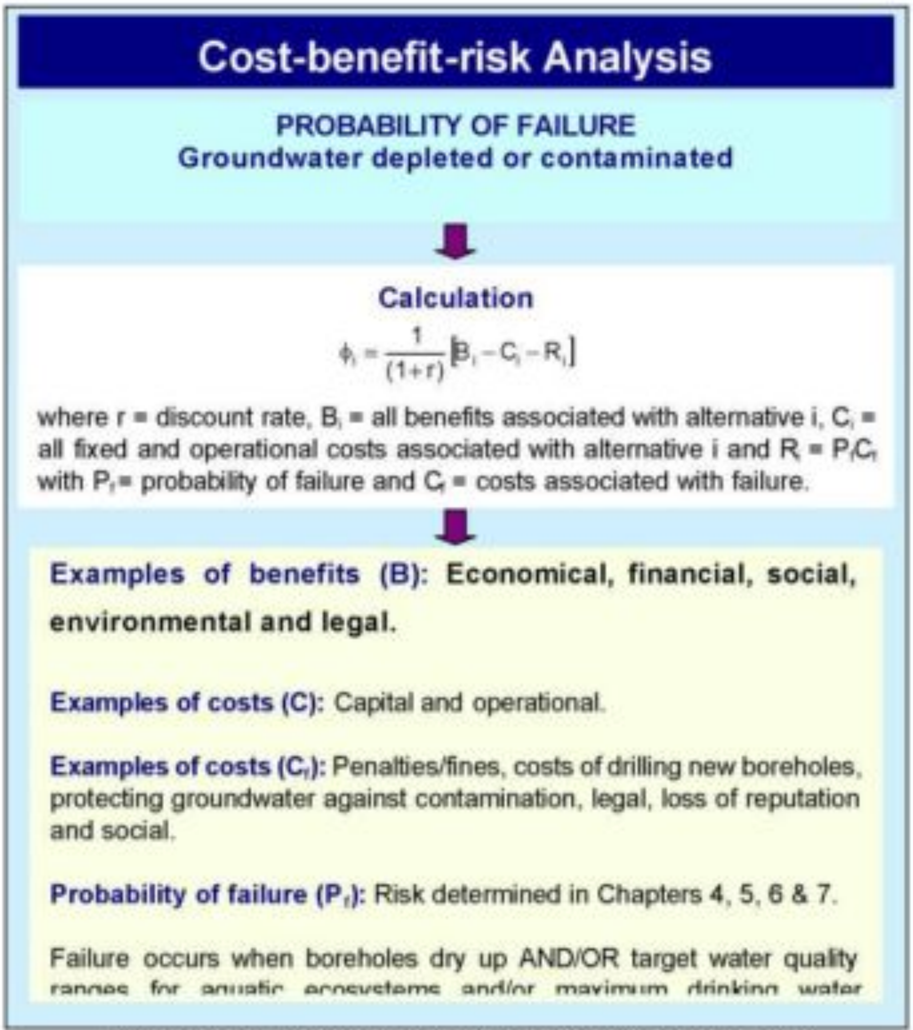


Figure 9-1. Methodology for Cost-benefit-risk analysis

9.3 EXAMPLE

When selecting "$" in the DT software the cost-benefit-risk analysis screen appears (Figure 9-2). The user can then decide what to include in the cost-benefit-risk analysis: the user can consider sustainability options, contamination options, health options, ecological options or any combination of the four. For each option (Figure 9-2) the user must enter the discount rate (interest %), all the costs involved and the probability of failure, which is obtained from performing a sustainable, contamination,

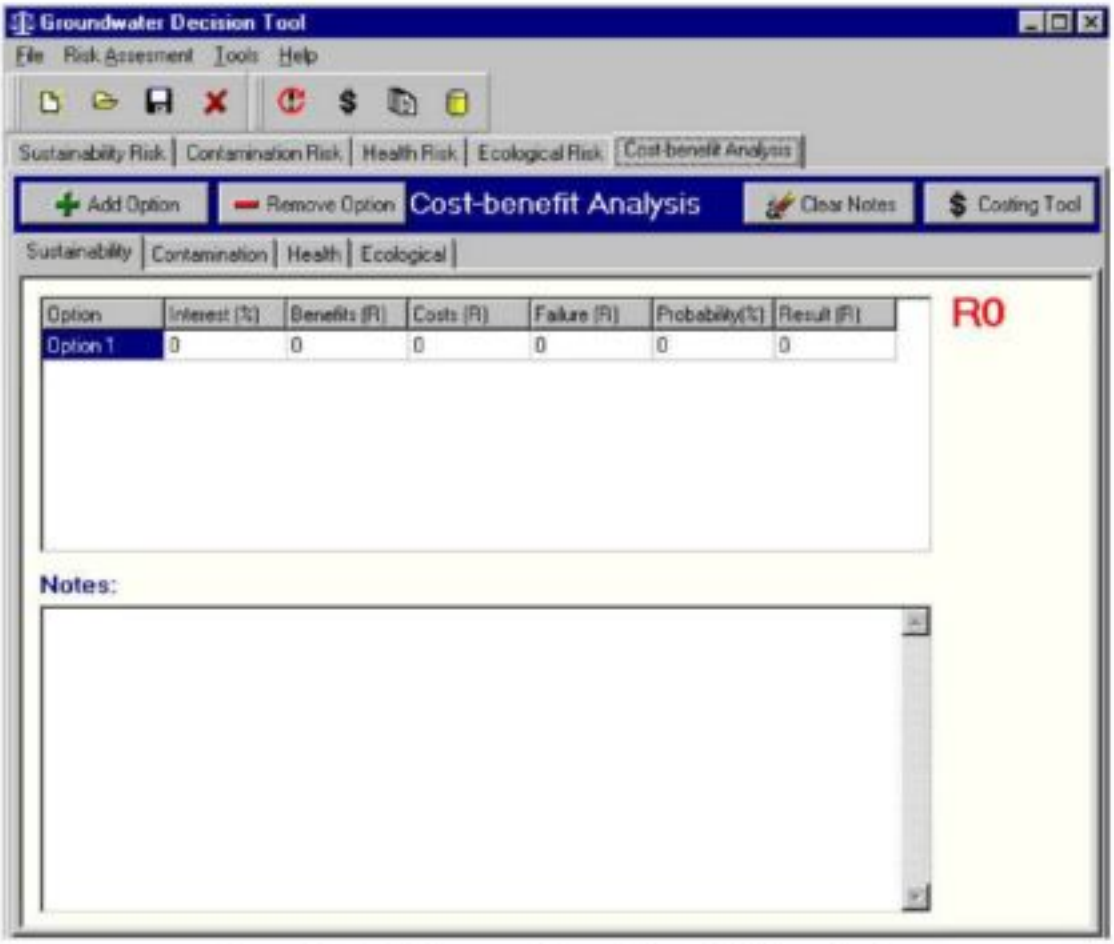


Figure 9-2. Initial screen for cost-benefit-risk analysis

health or ecological risk assessment. The DT will then calculate the final costs associated with the option. However by selecting "Costing Tool" the DT provides the user with a breakdown of what must be included in the costs. The DT will then calculate the final costs. In addition the user can enter which risk assessment results must be used, and the DT will automatically include the respective risk in the calculation. The "Notes" that occurs below the costing table allows the user to record information concerning each option. The DT does not optimise the objective function and the user must decide on the option best suited for the situation under investigation.

For example consider the hypothetical case study adapted from the one discussed by Rosen and LeGrand (1997) where a small storage facility for an organic compound is situated 200 m up-gradient from the property boundary (Figure 9-2). The facility contains one storage tank above ground containing 100 m^3 of the organic compound. Leaks have been found in similar tanks and the organic compound is highly soluble. The owner of the adjacent property is likely to take legal action if contamination occurs.

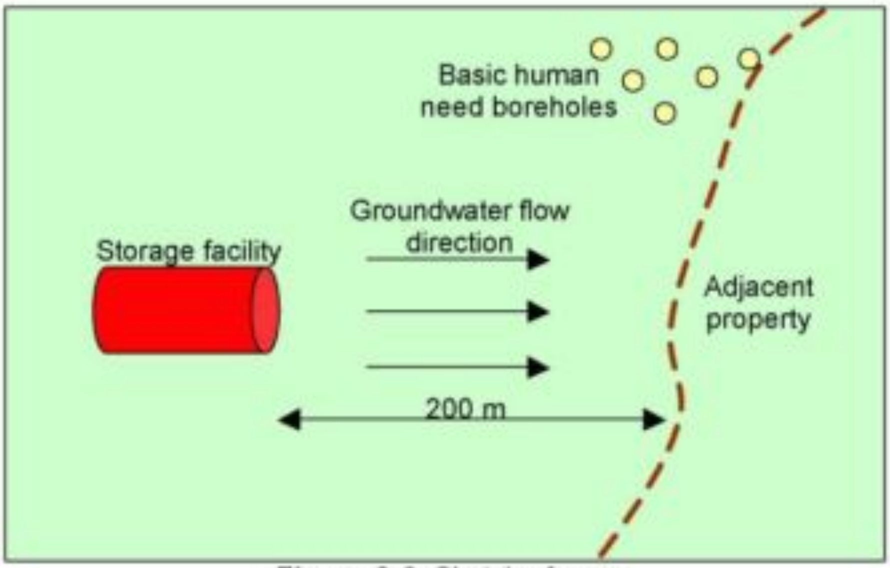


Figure 9-3. Sketch of area

Various contamination risk assessments were generated and the values used in the cost-benefit-risk analysis tool to determine the financial implications of each risk. The contamination risk assessment considered the possibilities of pollution entering the adjacent property. In addition there are approximately 6 basic human need boreholes in the area. Therefore a health risk assessment will have to be conducted for various scenarios to determine the health impacts at the basic human need boreholes.

As the health risk assessment and contamination assessment have been discussed in detail they will not be included here, hypothetical risks will be assumed for each scenario. The scenarios, included in the cost-benefit-risk analysis, are summarised in Table 9.1.

Table 9.1: Summary of scenarios used in cost-benefit-risk analysis

Scenario	Description	**Associated risks (%)**	
		Health	**Contamination**
1	There is a leak in the tank and the owner does not take any preventative measures. The owner also refuses to remediate the plume.	70	99
2	The owner builds a cutoff wall along the boundary of the property.	99	10
3	There is a leak in the tank and the owner does not take any preventative measures. However he is prepared to pay for the remediation.	70	99

The risks and associated costs of each of these scenarios are included in the cost-benefit-risk analysis. The contamination cost-benefit-risk values for scenario 1 are shown in Figures 9-4 and 9-5.

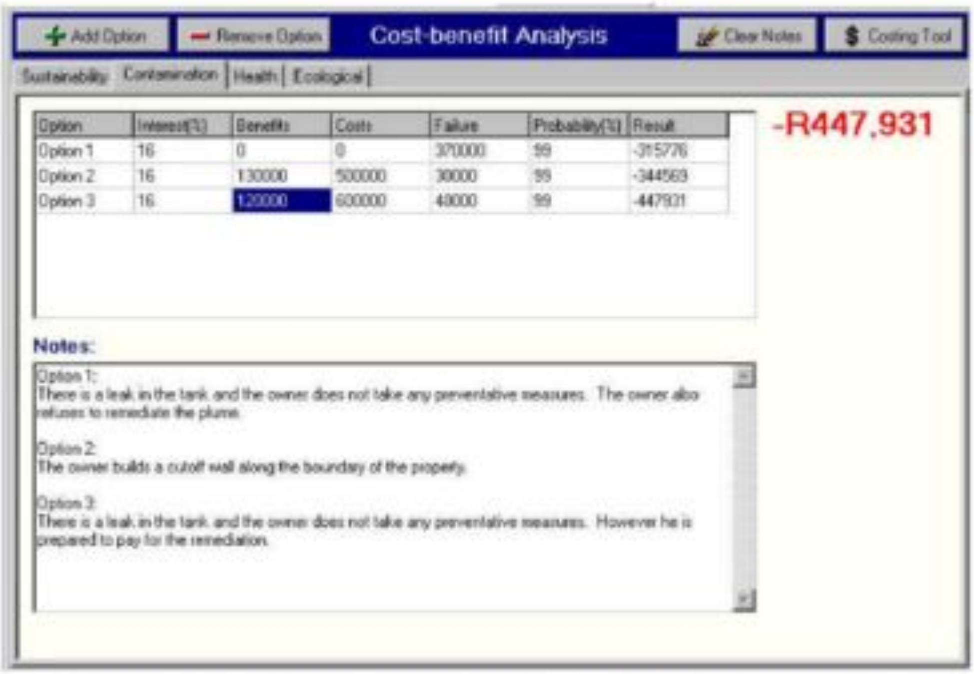


Figure 9-4. Contamination cost-benefit-risk analysis for scenario 1

Figure 9-5. Costing tool for contamination cost-benefit-risk analysis for scenario 1

The DT tool can also generate a report. No reports have been generated in any of the examples discussed in this report, however a report can be generated for all of them. To demonstrate the generation of a report, the results from the above scenario will be shown in report format.

When selecting  the report generator will appear. The initial screen of the report generator can be seen in Figure 9-6. The user can given the report a title and select the information he wishes to include in the report.

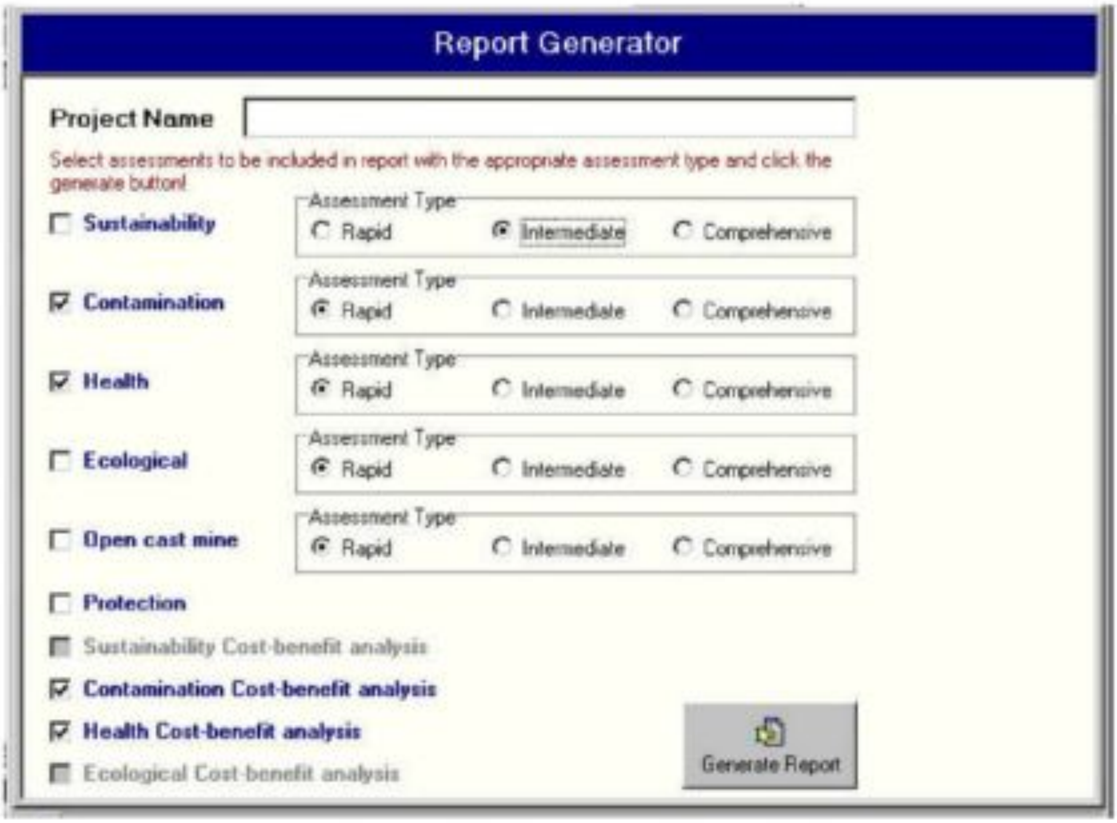


Figure 9-6. Initial screen of report generator

In this example the results of the contamination and health cost-benefit-risk analysis have been selected. The relevant sections of the report are shown below.

Health Cost-benefit Analysis

Option 1:
There is a leak in the tank and the owner does not take any preventative measures. The owner also refuses to remediate the plume.

Option 2:
The owner builds a cutoff wall along the boundary of the property.

Option 3:
There is a leak in the tank and the owner does not take any preventative measures. However he is prepared to pay for the remediation.

Option , Cost (R)

Option 1 , -138793
Option 2 , -196293
Option 3 , -521552

Contamination Cost-benefit Analysis

Option 1:
There is a leak in the tank and the owner does not take any preventative measures. The owner also refuses to remediate the plume.

Option 2:
The owner builds a cutoff wall along the boundary of the property.

Option 3:
There is a leak in the tank and the owner does not take any preventative measures. However he is prepared to pay for the remediation.

Option , Cost (R)

Option 1 , -315776

Option 2 , -344569

Option 3 , -447931

CHAPTER 10

Discussion, Conclusions and Recommendations

10.1 GENERAL

Water supply of acceptable quality is necessary for the improvement of the quality of life, and is essential in the maintenance of all forms of life. Limited water resources in South Africa have led to more emphasis being placed on groundwater. This report introduces a risk-based decision tool to be used for the management of groundwater.

A risk can be defined as the probability that an adverse event will occur under specified circumstances. Effective decision-making involves the management of risks: identifying, evaluating, selecting and implementing actions to reduce risk. Risk assessment is a technique that provides such information to the manager, thereby facilitating the complex and integrated decisions necessary.

In order to obtain accurate results from the risk assessment process, accurate data must be used. This report sets aside a chapter to discuss both aquifer and contaminant parameters and methods to obtain both sets of parameters.

The DT is divided into three tiers namely a rapid, intermediate and comprehensive assessment. For each of the tiers the following risk assessments can be performed:

- A groundwater risk assessment can be defined as the probability of an adverse effect or effects on the sustainability and/or quality of groundwater associated with measured or predicted hazards.
- A groundwater health risk assessment can be defined as a qualitative or quantitative process to characterise the probability of adverse health effects associated with measured or predicted levels of hazardous agents in groundwater.
- Ecological risks of interest differ qualitatively between different stresses, ecosystem types and locations. A groundwater ecological risk assessment quantifies the impacts of groundwater quantity and quality on ecosystems.

Once the desired risk assessments have been completed, cost-benefit-risk analyses can be used to aid in decision-making regarding the management of a groundwater resource.

A cost-benefit-risk analysis is defined as a set of procedures used for defining, comparing and measuring benefits and costs, which originate from either an investment or the operation of an activity.

Since the early 1980's geohydrologists and engineers have developed a number of techniques for protecting groundwater. Protection is divided into two categories: measures to prevent failure and pollution of water resources, and measures to remedy the effects of polluted water resources.

On completion of the different aspects of the DT a report will be generated including the input data and the results of the risk assessments and cost-benefit-risk analysis. Depending on the user, prevention measures can be included. Unfortunately no in-depth study has been completed on remediation options, but the user will be able to browse through the various options.

The information acquired from the DT can be used for risk management.

10.2 INTERPRETATION OF RISK

Risk assessment is a way of thinking about or analysing a situation, and as such it is a combination of science and judgement. Risk is a combination of two factors: (1) the chance that an adverse event will occur and (2) the consequences of that event. In this report there are four different risk assessments namely:

- Risk of a borehole or groundwater resource failing.
- Risk of a groundwater resource being contaminated.
- Risk of poor groundwater quality affecting human health.
- Risk of an aquatic ecosystem being affected by changes in groundwater quantity and/or quality.

Risk values are stated as a percentage. The higher the percentage the greater the potential of negative impacts. The highest risk obtainable is 99% indicating that under the conditions stipulated in the respective risk assessment there chances of the agent (be it groundwater, human health or an aquatic ecosystem) being impacted are extremely high.

It is the manager's decision as to whether a risk is acceptable or not. This decision must be taken considering both legislation and affected parties. For example a manager might decide a 25% chance of a borehole failing is acceptable, however a 25% chance of a person becoming seriously ill when drinking contaminated groundwater is not acceptable.

The calculated risks are dependent on the confidence in data and method used to calculate the risks, therefore it is important for the manager to understand the fuzzy logic methodology and the associated membership functions.

Risk management is the process of identifying, evaluating, selecting and implementing actions to reduce risks.

10.3 CONCLUSIONS

The DT developed in this report relies heavily on the expertise of geohydrologists, assumptions and approximations of real world conditions. Together with the heterogeneities present in groundwater systems it is impossible to guarantee the accuracy of the methodologies and the reader must take this into consideration. However as Hurst (1957) stated: *It is usually better to do something which is 95% effective immediately, rather than to wait several years to improve the solution by 4%.*

The DT can be a useful tool for a groundwater manager to use in order to obtain an understanding of the groundwater situation in a particular area and the impacts thereof. In addition the DT can be used to rank groundwater related problems, thereby making groundwater management and protection an achievable task.

10.4 RECOMMENDATIONS FOR FURTHER RESEARCH

The DT presents a fuzzy logic based method to do risk assessments concerning groundwater. Included are methodologies to characterise fractured rock aquifers. There is ongoing research concerning these aquifers and as new methodologies are developed it is important to include them in the DT.

This DT has been developed over a period of two and half years and even though it has been tested and calibrated by experts, it is important to note that in order to obtain more accurate results, it must be validated over a period of many years.

In addition the database of the DT has been populated with information; it can be expanded and more detail can be added.

The ecological risk assessment is limited to a few indicators to determine the risks for aquatic ecosystems. This assessment can be developed to include aspects such as flow conditions in rivers and fish species. In addition the impacts of groundwater on terrestrial ecosystems need to be considered and included in the ecological risk assessment. Accommodating these factors complicates ecological risk assessments.

The cost-benefit-risk analysis is crude and this can be developed into more comprehensive computations such as those discussed by Janse van Rensburg (1992).

Even though uncertainty has indirectly been included in the DT. Further development of the DT should include a comprehensive uncertainty analysis. The uncertainty analysis should include aspects such as the quality of data, the relationships between potential hazards and effects of concern and, the methods used to calculate risks. The uncertainty analysis thereby highlights the limitations of the risk assessment allowing decisions to be made in a more transparent fashion.

CHAPTER 11

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APPENDIX A

Methods to Analyse Tracer Tests

A1. ANALYSING TRACER TESTS

Tracer tests aim to relate the concentration of chemical, biological or solid substances measured in observation boreholes to the flow velocity. Because tracer tests under natural conditions with several observation boreholes are time- and cost-expensive, different single-well and dual-well tracer tests were explored. Both single-well and multiple-well tracer tests are described together with their analysing methods in the following sections.

Single-well Tracer Tests, Natural Gradient

As the name indicates, single-well tracer tests are conducted in one borehole only, meaning that injection of the tracer and measurement of the concentration take place in the same borehole. Conducting and measuring types vary for the different test types.

Point Dilution Test

The single-well point dilution test method aims to relate the observed rate of tracer dilution in a borehole (Freeze and Cherry, 1979), or in a segment isolated in a borehole, to the average groundwater velocity in the aquifer. The groundwater through flow gradually removes a tracer introduced to the well from the well bore to produce a time-concentration relationship, from which the Darcy velocity is computed. The tracer is not recovered by pumping. Under conditions of steady flow and thorough mixing of the tracer in the borehole, the Darcy velocity is computed from a dilution test as (Drost and Neumaier, 1974):

$$q = \frac{W}{\alpha At} \ln\left(\frac{C_0}{C}\right)$$

With:

W = volume of fluid contained in the test section

A = cross sectional area normal to the direction of flow

C_0 = Tracer concentration at t = 0

C = Tracer concentration at time = t

α = Borehole distortion factor (between 0.5 and 4; =2 for an open well)
(note that $q\alpha = v^*$, where v^* = apparent velocity inside well)

t = Time when concentration is equal to C

In practice either the radial flow solution (porous media) or the parallel plate model (single fracture) is used to estimate the cross sectional area A:

$$A = \pi r_w d \quad \text{(for the radial flow model)}$$

With:

r_w = well radius

d = the length of the tested section in the borehole

And $A = \pi r_w(2b)$ for the parallel plate model

Where

2b = equivalent aperture of the fracture.

The theory of equivalent aperture applied to tracer test data seems unreliable. But as it is possible to account for the flow geometry in generalising the flow in fractured aquifers, it is also possible to estimate the cross-sectional area *A* in a more general way by using fractional flow dimensions.

In the case of the dilution test, the cross sectional area for n-dimensional flow is thus given by (half the borehole circumference is used):

$$A = \frac{r_w^{\,n-1}\pi^{\frac{n}{2}} b^{3-n}}{\Gamma(\frac{n}{2})}$$

If n=2, the equation reduces to $A = \pi r_w d$ (describing radial flow) and for n=1, the equation describes linear flow (analogue to the parallel plate model). For non-integer values of n, the flow dimension becomes a fractional dimension.

Injection-Withdrawal Test

The single well injection-withdrawal test (also known as the drift and pumpback test) was described by Borowczyk *et al.* (1966) and Leap and Kaplan (1988) for a homogeneous, isotropic and confined aquifer. To conduct a single well injection-withdrawal test a tracer is introduced to the standing water column of the test well and allowed to drift, under natural gradient, away from the well bore. After a period of time pumping the test well retrieves the tracer plume. Groundwater flow velocity is then calculated, based on the amount of pumping needed to recover the tracer, by the following equation (Leap and Kaplan, 1988):

$$v = \frac{\sqrt{Qt_p / \pi \varepsilon D}}{t_d}$$

where:

v = seepage velocity (m/d)

Q = Pumping rate during recovery of tracer (m^3/d)

t_p = Time elapsed from start of pumping until the centre of mass of the tracer is recovered (d)

ε = Kinematic porosity

D = Aquifer thickness (m)

t_d = Time elapsed from the injection of tracer until the centre of mass of the tracer is recovered (d)

The inconsistency of this method is in the assumptions. The method should "account for regional velocities that are too high to be neglected during pumpback", but one of the assumptions is that "the aquifer is homogeneous and isotropic and possesses no regional hydraulic gradient" (Leap and Kaplan, 1988).

Neglecting the natural hydraulic gradient during the pumpback phase will yield the equation of Borowczyk *et al* (1966):

$$v = \frac{(Q/\Pi\varepsilon\, b)^{1/2}\, t_p^{1/2}}{t_*}$$

where

t_p = Time elapsed from start of pumping until the centre of mass of the tracer is recovered (d)

t_* = drift time of tracer ($t_p - t_i$).

The equations of Leap and Kaplan (1988) and Borowczyk *et al.* (1966) are applicable in the case where a flow dimension n of 2 exists, i.e. steady state horizontal flow during drift phase and radial flow during the pumping phase. It is also possible to estimate the seepage velocity in a more general way by using fractional flow dimensions (Barker, 1988). The generalised equation of Leap and Kaplan reads:

$$v = \frac{(Qt_p / \varepsilon\, b^{3-n}\beta_n)^{\frac{1}{n}}}{t_d}$$

If n=2 (meaning radial flow field), the equation reduces to the original equation of Leap and Kaplan (1988). Using the drift time t_* instead of t_d generalises the equation of Borowczyk *et al.* (1966), which then reads:

$$v = \frac{(Qt_p / \varepsilon\, b^{3-n}\beta_n)^{\frac{1}{n}}}{t_*}$$

Hall *et al.* (1991) uses a combination of the Leap and Kaplan equation and Darcy's Law ($v = Ki/\varepsilon$) to estimate the groundwater velocity and kinematic porosity with the same assumptions like Leap and Kaplan (1988):

$$V = \frac{Qt_p}{\pi D t_d^2 Ki}$$

and

$$\varepsilon = \frac{\pi D K^2 i^2 t_d^2}{Qt_p}$$

Where:

V	=	seepage velocity
Q	=	Pumping rate during recovery of tracer
t_p	=	Time elapsed from start of pumping until the centre of mass of the tracer is recovered
D	=	Aquifer thickness
t_d	=	Time elapsed from the injection of tracer until the centre of mass of the tracer is recovered
K	=	Horizontal hydraulic conductivity
i	=	Horizontal hydraulic gradient
ε	=	Kinematic porosity

For the case of a non-integer flow dimension of n, the Hall equations generalise to (for n>1):

$$V = \left[\frac{Qt_p}{\beta_n b^{3-n} t_d^{\,n} Ki}\right]^{\frac{1}{n-1}}$$

and

$$\varepsilon = \left[\frac{\beta_n b^{3-n} (Kit_d)^n}{Qt_p}\right]^{\frac{1}{n-1}}$$

Multiple-well Tracer Tests, Forced Gradient

The most common methods in conducting tracer tests are multiple-well tracer tests, where a specific flow field is created by recharging and / or abstracting water at different boreholes. Injection and observation of the tracer will then take part at specific places in the created flow field. The tests and the analysing methods differ according to the created flow field.

Radial Convergent Test

Pumping a well until steady state conditions are reached creates a radial convergent flow field. A tracer is then quickly introduced in an injection well located in the vicinity of the pumping well in such a way that a minimum disturbance of the flow field is caused, while the tracer breakthrough curve is monitored at the pumping well. Analyses of the resulting breakthrough curves yield estimates of the kinematic porosity, aquifer dispersivity and groundwater velocity. The convergent test is attractive because it is theoretically possible to recover the tracer from the aquifer completely. Furthermore, it most closely represents reality as groundwater pollution often occurs in the vicinity of pumping wells where radial flow fields are present. The convergent tracer test, in combination with the borehole dilution test, has proved to be a powerful hydrogeological tool for measuring groundwater velocity and kinematic porosity.

When the tracer test is conducted in a homogeneous and isotropic aquifer, where the transport is dominated by advection and dispersion and where molecular diffusion can be neglected, the approximate solution for a converging radial flow with a pulse injection is given by (Sauty, 1980):

$$c(r,t) = \frac{\Delta M}{2Q\sqrt{\pi D_L t^3}} \exp\left[-\frac{(r - vt)^2}{4D_L t}\right]$$

with:

ΔM	=	injected mass of tracer per unit section (Mass (kg)/ Thickness (m))
D_L	=	longitudinal dispersion coefficient (m^2/s); $D_L = \alpha_L v$
α_L	=	longitudinal dispersivity (m)
v	=	v_f; groundwater velocity under forced gradient (m/s)
Q	=	pumping rate of the well (m^3/s)
r	=	radial distance (m) between the two boreholes

Since matrix diffusion cannot be neglected in most cases of solute transport in fractured media, the analysing method for tracer tests should account for this effect. A model and analytical solutions were developed to estimate velocity, dispersivity and matrix diffusion from radial convergent tracer tests:

$$c_f(t) = \frac{Ma}{2\pi Q}(Pet_0)^{1/2} \int_0^t \exp\left[-\frac{Pe(t_0 - u)^2}{4ut_0} - \frac{a^2u^2}{t-u}\right] \frac{du}{\left[u(t-u)^3\right]^{1/2}}$$

where t_0, a and Pe are the fitting parameters, as defined below.

t_0	=	x_0/v, mean travel time of mobile water in the fracture
x_0	=	distance along the direction of flow in the fracture between the entrance to the system and the observation point
u	=	$(Pe\ t_0)/(4\xi^2)$
ξ	=	integration variable
Pe	=	$v\ x_0 / D = x_0 / \alpha$, Peclet number
D	=	coefficient of intrinsic dispersion
α	=	intrinsic dispersivity

a = $n_p (D_p R_{ap})^{1/2} / (2b)$, diffusion parameter

n_p = matrix porosity

D_p = molecular diffusion coefficient in the matrix

2b = fracture aperture

R_{ap} = Retardation factor caused by exchange in the matrix

A2. SOFTWARE FOR ANALYSING TRACER TESTS

Two software programmes were developed for conducting and analysing tracer tests in fractured aquifers. Both are prepared as easy to handle spreadsheets in MS EXCEL, written in Visual Basic. While the programme TRACER-PLAN can be seen as an expert system supporting the user in planning the tracer test, the programme TRACER consists of several tools for analysing the tracer test, applying either the standard methods or the approach of non-integer flow dimension.

TRACER-PLAN

The programme TRACER-PLAN consists of different tools to support the user in choosing the correct equipment and planning the best way of conducting the test. The different parts and spreadsheets of the programme are briefly described below.

Basic Information

The basic information, needed for the program to calculate and recommend the parameters for the test set-up, have to be inserted in this screen. The following information is required, either as known or assumed values:

- Purpose of testing (i.e. which parameters are to be estimated)
- Geological structure
- Depth of fracture zone, or zone of interest
- Thickness of fracture zone
- Hydraulic parameters of formation and / or fracture
- Available boreholes on test site, including distance from each other

The suggestions based on this information are given at the bottom of the screen and transferred to the following screens, 'Test Set-Up' and 'Simulation'.

Test Set-Up

Based on the information on the first screen, the test set-up and the necessary equipment are suggested. However, the suggestions can be changed and adapted to the situation on the test site. After fixing the equipment, the program calculates and / or recommends the following parameters for conducting the test:

- Flow rate for circulating
- Circulating time
- Injection time for injecting tracer into the borehole
- Kind of tracer
- Amount of tracer
- Flow rate for abstraction, if applicable

The two latter parameters are calculated from both the test set-up and the forward simulation, as described in the next section.

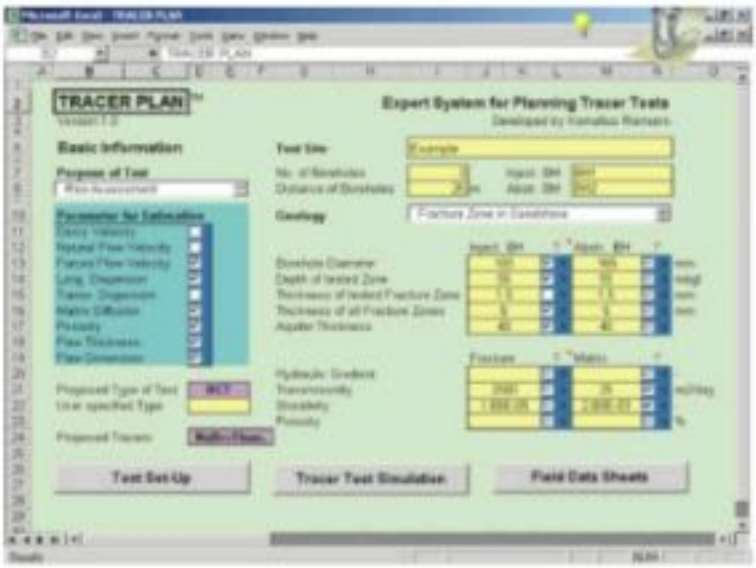


Figure 1 Input Screen 'Basic Information' of software TRACER-PLAN

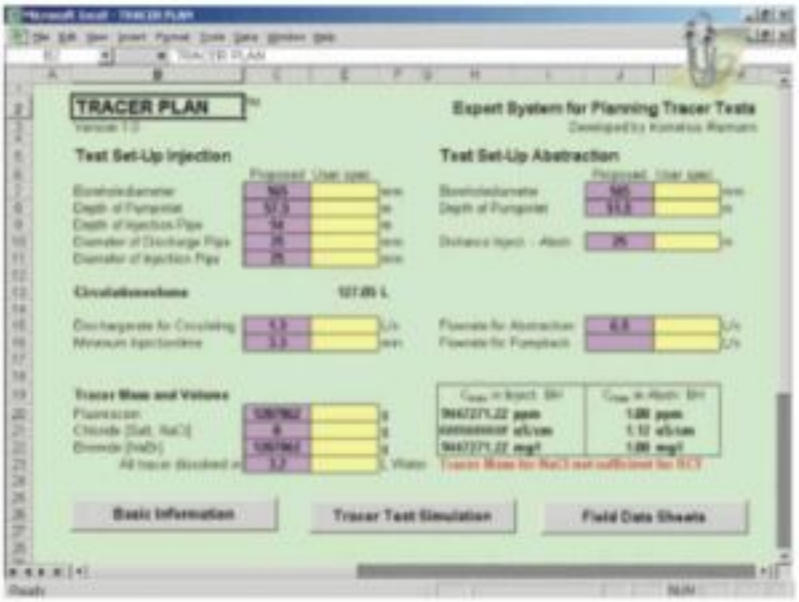


Figure 2Input Screen 'Test Set-Up' of software TRACER-PLAN

Simulation

Using the available data about the flow system and the test set-up a forward simulation of the tracer test is produced before conducting the test to estimate the parameters, needed for conducting the test. The result of the simulation will feed back automatically to the above-mentioned parameters and gives new recommendations for their range of suitable values. For example, if the simulation shows that the amount of tracer is too small to be detected in the abstracted water, a backward calculation yields a recommended value for the amount of tracer for injection.

During the test the simulation can be used to optimise the test set-up and the initial parameters. For instance after the dilution part is completed a rapid analysing of the test data will yield a first approximation of the Darcy velocity, which will result in a better simulation of the withdrawal part, if inserted in the program as a known value.

The simulation accounts for the uncertainty of the input data by means of applying the upper or lower value of the possible range. If a parameter value is known a priori, the uncertainty is set as small, while assumed parameter values will get an uncertainty range of up to one order of magnitude (lognormal distribution) or 50% (normal distribution).

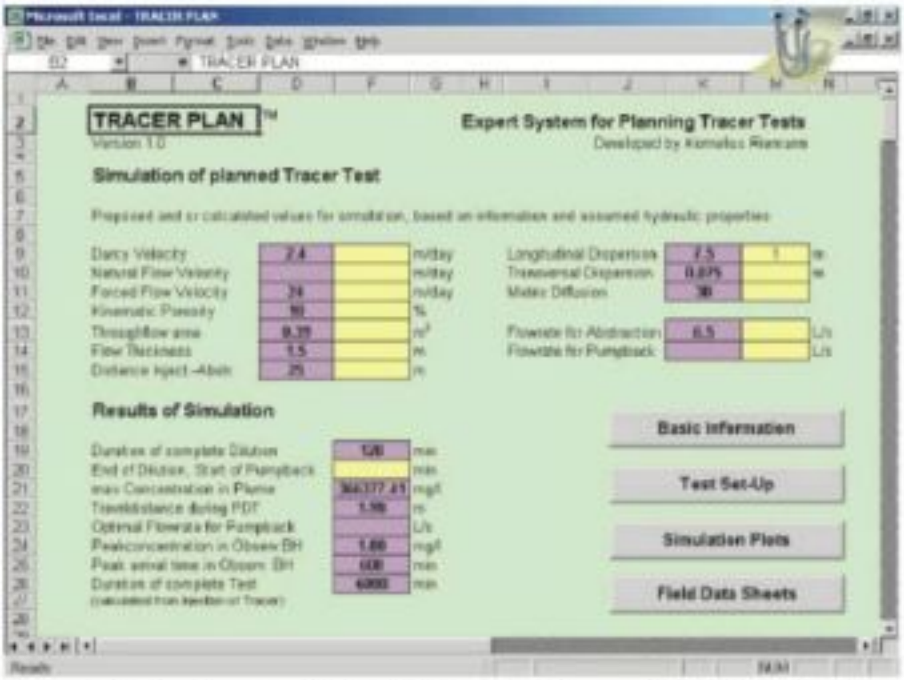


Figure 3Output screen 'Simulation' of software TRACER-PLAN

TRACER

In the following section an approach of non-integer flow dimension is proposed to analyse hydraulic and tracer test data. Since the method of non-integer flow dimension, applied to tracer test data, is a new development, no existing software programme can handle this approach. Therefore the programme TRACER was developed, mainly to account for the non-integer flow dimension when analysing tracer tests. The different parts and spreadsheets of the programme are briefly described below.

In general the structure is equal in all parts of the program and further information is given in comments related to the input cells.

- Input cells are marked yellow
- Linked cells are marked light blue (i.e. information required on other screen)
- Calculated cells are marked grey (i.e. never overwrite!)
- Results are marked in different colours

Main

The main screen (Figure 4) serves as an entering platform for the program. All relevant parts of the program can be opened from this screen. Furthermore the basic information about the conducted test or tests is required in this screen. The three options are:

- Forced Gradient test between 2 boreholes (RCT)
- Natural Gradient with Dilution and Withdrawal
- Natural Gradient just Dilution

Additional information is required, whether pumping test data from the same boreholes are available, which method for estimating the parameter flowdimension and porosity should be applied and about the names of abstraction, injection and observation boreholes, because this information will be used in the following parts of the program.

Hydraulic Test Data

The part of the program for the hydraulic test data is taken from the FC-program (Van Tonder *et al.*, 2001) and can be divided in the input of the field data, the diagnostic and the analysis. Because the focus of the program is analysing tracer tests, the analysing procedure of the hydraulic test data is reduced to the necessary part of estimating the flow dimension and flow domain, applying the methods of Barker (1988).

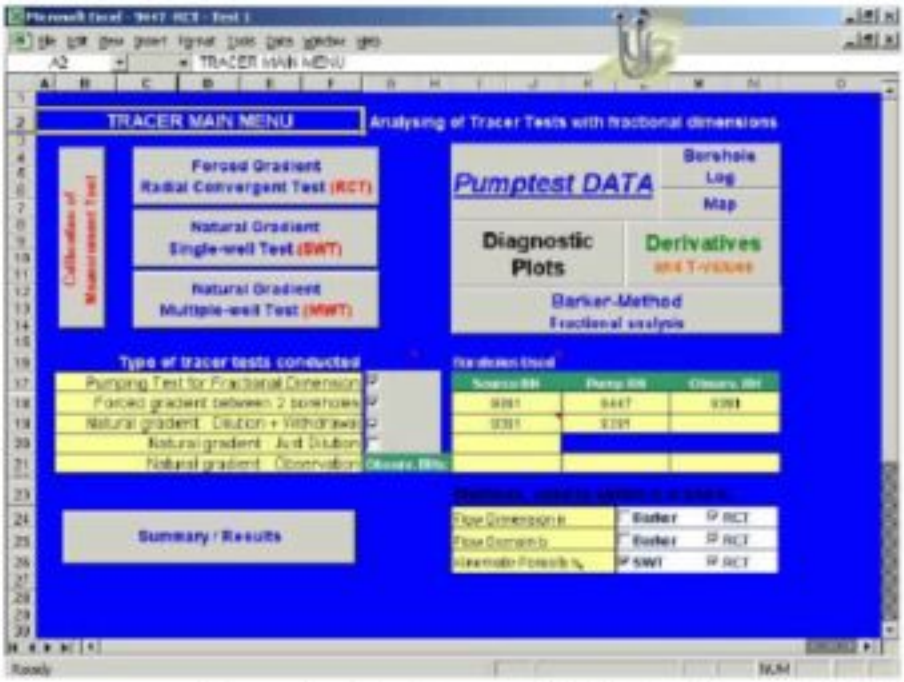


Figure 4 Input screen 'Main' of software TRACER

<u>Field Data (Pumptest Data, Borehole Logs, Map)</u>

The required data from the hydraulic test are the discharge rate Q in L/s, the time, elapsed since starting abstraction in minutes, the static water level and the measured water level in meter below ground level [mbgl] during the drawdown and recovery phase. The data sets for both the abstraction and observation borehole should be completed on this sheet.

For a preliminary estimation of the transmissivity and storativity the effective borehole radius of the abstraction borehole is required. The geographic data of the boreholes, such as x- and y-values and elevation are required in the screen 'map', where additionally there is the possibility to insert a bitmap with the borehole locations. The borehole logs can be inserted onto the screen 'Borehole Log' for the convenience of the user.

Diagnostic Plots

Several diagnostic plots are provided to enable the user distinguishing the different flow phases occurring during the test.

Derivatives and T-Values

The diagnostic tool of derivatives is separated due to their importance. The preliminary estimation of the transmissivity, based on the order of the first derivative, is included. Unlike the FC-program the tools are provided for both the abstraction and observation borehole.

Barker – Method, Fractional Analysis

The estimation of the flow dimension and flow domain, as obtained from hydraulic test data, is the crucial part of the analysing procedure. A non-linear least square method is implemented, which can be used manually or automatically, using the EXCEL-tool Solver. Due to the non-uniqueness of the method, additional information can be used as upper or lower bounds for the range of a single parameter. On the graph the field data and the simulated drawdown curve, applying the chosen parameter values, are plotted together for comparison.

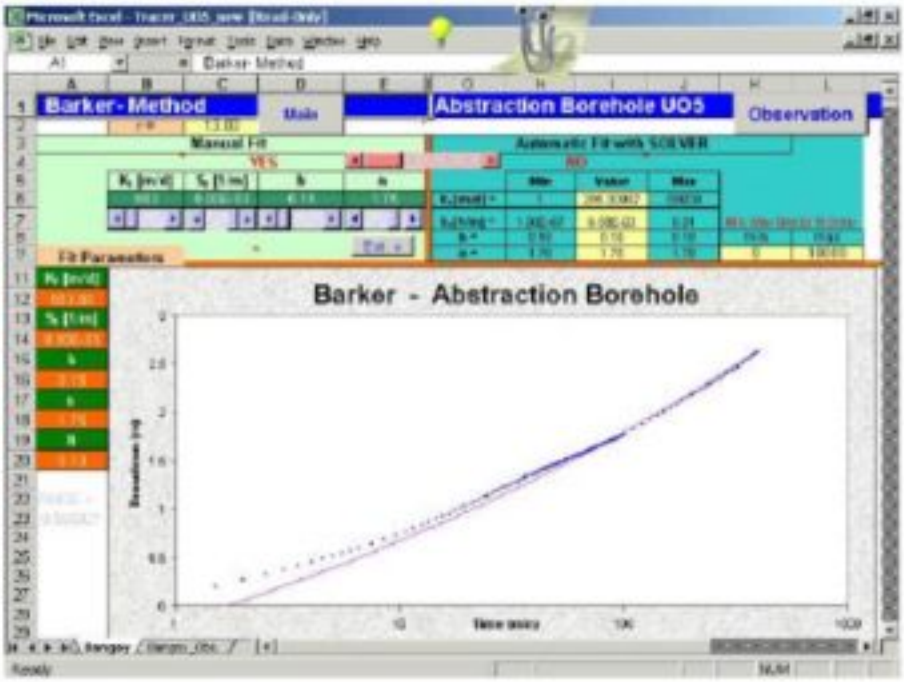


Figure 5 Input and output screen 'Barker' of software TRACER

Tracer Test Data

The tracer test data and analysing procedures are separated for each type of test. In the first step virtual calibration of the measurement tool is required, if the used tool cannot be calibrated in the field.

Calibration of Measurement Tool

The calibration screen provides the input of calibration data from laboratory comparison. Therefore the used measurement tool should be measured against fixed concentrations of different solutes at different scales, and the readings should be entered into the table. The result of the comparison is shown on the plot 'measured values vs. fixed values'.

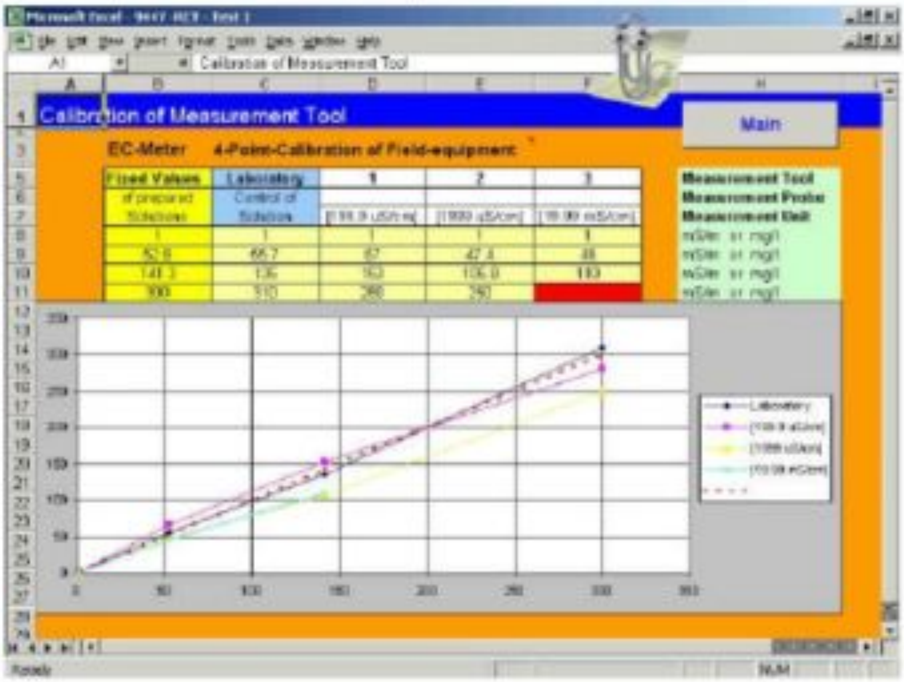


Figure 6 Input screen 'Calibration' of software TRACER

Forced Gradient, Radial Convergent Test (RCT)

The tracer test data and analysing procedure for a radial convergent test is divided into two parts, namely the injection (i.e. Point-Dilution test), and the abstraction (i.e. Radial Convergent test).

For analysing the injection part as a Point-Dilution test the field data and the test set-up are required. In detail:

- Radius of borehole,
- Interval length (i.e. distance from pump inlet to injection pipe outlet)
- Distortion factor (equal to 2 in open boreholes)
- Measurement unit (choose from selection)
- Measurement tool (choose from selection)

- Background value
- Time elapsed since start of injection
- Concentration in circulating water at time t
-
- Depending on the chosen approach for the analysing procedure (see 'Main' screen), the flow dimension, the flow thickness or the fracture aperture and the kinematic porosity will be calculated. However, the calculated values can be changed manually for comparison purposes. The field data are plotted in a semilog plot (concentration vs. time) and should show a straight line. Fitting the straight line will yield the Darcy velocity. The value for the forced flow velocity depends on the kinematic porosity and should be equal to the estimation from the radial convergent test.

Figure 7 Input and output screen 'Point Dilution' of software TRACER

The analysing procedure for the breakthrough curve in the abstraction borehole requires additional data about the established flow field:

- Abstraction rate (value is taken from 'Pumptest Data' sheet)
- Distance between injection and abstraction (value is taken from 'Map' sheet)
- Injected mass of tracer in kg
- Measurement units (chosen from selection)
- Time elapsed since injection of tracer in minutes
- Concentration of tracer in abstracted water at time t

On the graph the measured concentration is plotted against the elapsed time. Furthermore the relative mass recovery is calculated and plotted vs. time. Changing manually the values for the flow thickness, dispersion and velocity until the measured data and the simulated data show a good fit, will yield the estimation of the fitting parameters. The accuracy of the fit can be controlled by checking the calculated RMSE, which should be as small as possible.

With the combination of the Point-Dilution test and Radial Convergent test the kinematic porosity on the flow path and the flow dimension is calculated. These parameter values and the estimated flow thickness should be used when analysing the Point-Dilution test, as it is set as default in the program.

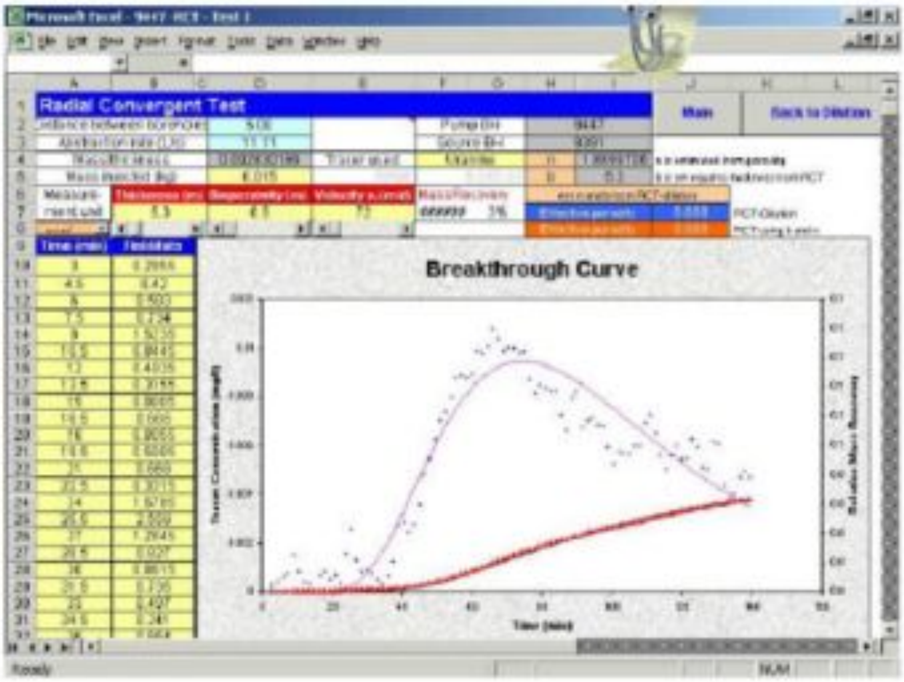


Figure 8 Input and output screen 'Radial Convergent' of software TRACER

Natural Gradient, Single-well Test (SWT)

The tracer test data and analysing procedure for a single-well test is divided into two parts, namely the injection (i.e. Point-Dilution test), and the abstraction (i.e. Injection-Withdrawal test). The injection part is equal to the injection part of a radial convergent test and described above.

For analysing the withdrawal part of a single-well tracer test the following information and data are required:

- Abstraction rate during withdrawal part
- Time of the drift phase (i.e. between injection and pumping)
- Time elapsed since start of pumping in minutes

- Concentration of tracer in abstracted water at time t
-
- Depending on the chosen approach for the analysing procedure (see 'Main' screen), the flow dimension, the flow thickness and the kinematic porosity will be calculated. However, the calculated values can be changed manually for comparison purposes. The measured concentration is plotted against the elapsed time and should show a breakthrough curve. Moving the purple dot to the centre of recovered mass, which is shown by the minimum value of 'delMass', yield the flow velocity. For comparison both the approach of Leap and Kaplan (1988) and the method of Borowczyk *et al.* (1966) are included.

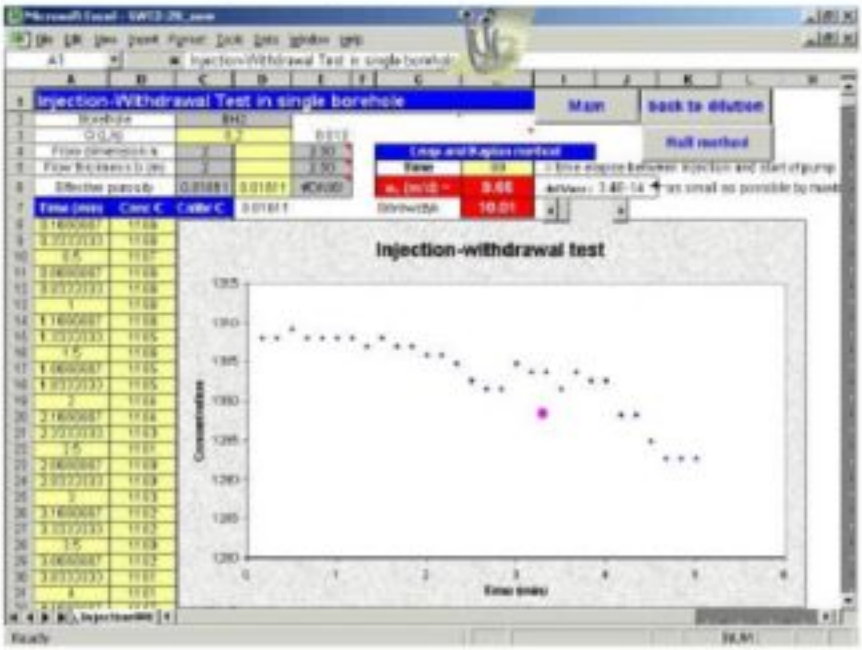


Figure 9 Input and output screen 'Injection Withdrawal' of software TRACER

Natural Gradient, Multiple-well Test (NFT)

The tracer test data and analysing procedure for a natural flow test is divided into two parts, namely injection (i.e. Point-Dilution test) and observation. The injection part is equal to the injection part of a radial convergent test as described above.

The analysing procedure for the breakthrough curves in the observation boreholes requires additional data:

- Approximated direction of flow in degree
- Relative position of observation boreholes (values are calculated from 'Map' sheet)
- Injected mass of tracer in kg

- Measurement units (chosen from selection)
- Time of injection (date and time)
- Time of measurements (date and time)
- Concentration of tracer in observation borehole at time t

The concentration in observation borehole 1, situated in the assumed flow direction, is plotted against time and compared with simulated curves, using one-dimensional and two-dimensional approaches. The concentrations in the other two boreholes are plotted against time and compared with simulated curves using a two-dimensional approach. Changing the values of the flow velocity, longitudinal dispersion, transversal dispersion and eventually flow direction manually until best fits of all breakthrough curves is reached, will yield estimations of the fitting parameters.

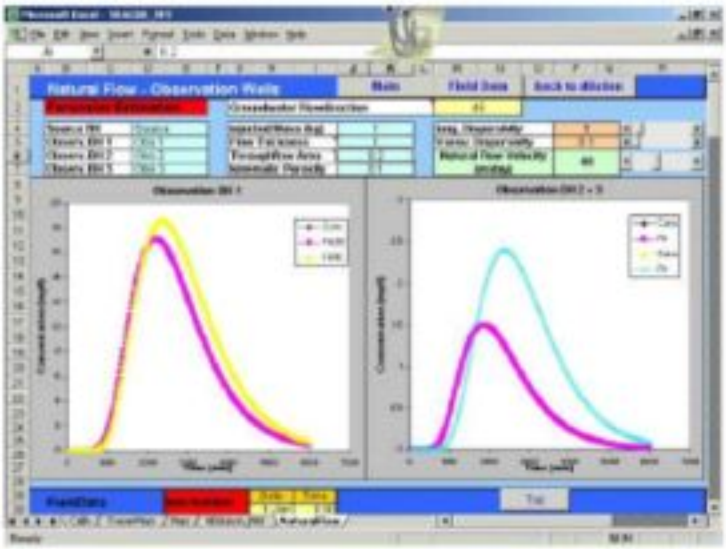


Figure 10 Input and output screen 'Natural Flow' of software TRACER

Results

The results of all analysing procedures and tests are summarised in the 'Result' screen. Additionally the following parameters are calculated, which are not implemented explicitly in the analysing sheets:

- Transmissivity of the formation, using the FC approach (i.e. T_{late})
- Hydraulic gradient during hydraulic and / or radial convergent tests
- Natural hydraulic gradient

Figure 11 Output screen 'Results' of software TRACER

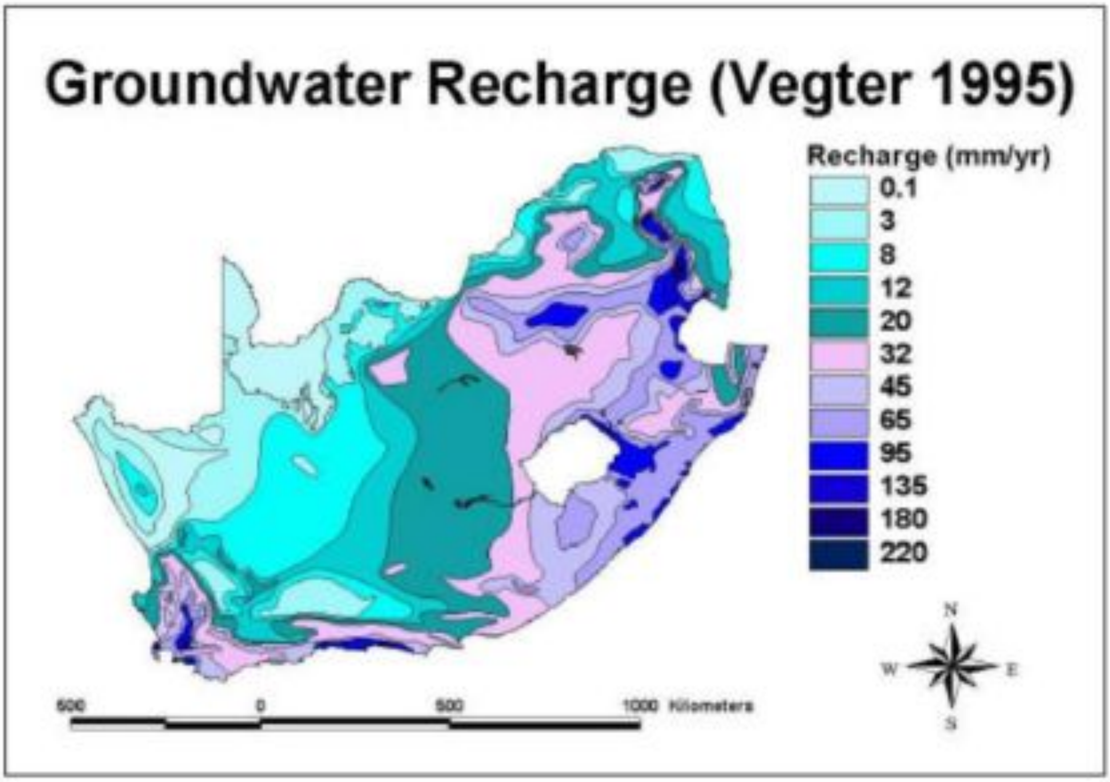
ALL SOFTWARE IS INCLUDED ON ATTACHED CD.

For more information concerning tracer tests refer to *New Developments in conducting and analysing tracer tests in fractured rock aquifers* written by K Riemann (2002), Institute for Groundwater Studies, University of the Free State, Bloemfontein.

APPENDIX B

Groundwater Sustainability Risk Assessment Information

B1. VEGTER'S GROUNDWATER RECHARGE MAP



B2. STORATIVITY VALUES AND AQUIFER TYPES

The storativity values given to the various aquifer types were determined by G van Tonder, and are listed in the table below:

Aquifer type	Storativity
Fractured hard rock	1×10^{-3}
Karoo fractured rock	3×10^{-3}
Table mountain	8×10^{-3}
Dolomite	1×10^{-2}
Porous	1×10^{-1}

B3. CALCULATIONS OF BOUNDARY CONDITIONS FOR THE INTERMEDIATE AND

COMPREHENSIVE SUSTAINABLE RISK CALCULATION *(Taken directly from Van Tonder et al., 1999)*

1. Extrapolation of Pumping Test Drawdown

The extrapolation of the drawdown of the pumping test is the sum of the drawdown that is due to the production well, s_{Well}, and the boundaries, $s_{Boundary}$:

Eq. 1: $$s(t = t_{long}) = s_{Well} + s_{Boundary}$$

The following sections distinguish between the extrapolation of s_{Well} and $s_{Boundary}$.

1.1 Extrapolation of Production Well Drawdown

The drawdown that is due to the production well is extrapolated by a Taylor series expansion around the late measurement points of the drawdown at $t \approx t_{EOP}$ (subscript EOP denotes end of pumping test). The Taylor series expansion is performed with respect to the logarithm of time, $\log_{10}$. A second order approximation is assumed to be sufficient:

Eq. 2:

$$s_{Well}(t = t_{long}) \approx s(t = t_{EOP}) + \frac{\partial s}{\partial \log t}\Big|_{t=t_{EOP}} (\log t_{long} - \log t_{EOP}) +$$
$$+ \frac{1}{2}\frac{\partial^2 s}{\partial (\log t)^2}\Big|_{t=t_{EOP}} (\log t_{long} - \log t_{EOP})^2$$

The time t_{EOP} must be large enough to ensure that the drawdown has already passed the early time flow behavior that is due to well bore storage, fracture flow and double porosity effects. This can clearly be monitored by looking at the derivative plot $\partial s / \partial \log t$. Usually the effect of the boundaries can only be seen at very late times of the pumping test. For simple geometries of the boundaries, image well theory can be applied to analyse the effect of the boundaries on the drawdown. This is shown in the following section.

1.2 Extrapolation of the Boundary Drawdown

Four simplified cases of no-flow boundaries are investigated:

- A single barrier boundary
- Two barrier boundaries intersecting at 90^0
- Two parallel boundaries
- A closed square barrier boundary

- **Single barrier boundary**

The single barrier boundary is illustrated in Fig.1:

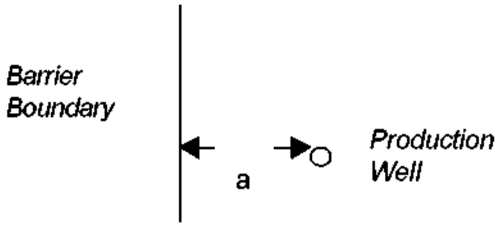


Fig. 1: Single Barrier Boundary

The influence of the barrier boundary can be described by constructing an image well. This image well is located on the other side of the boundary at the same distance as from the boundary to the pumping well. The drawdown in the pumping well due to the barrier boundary is expressed by

Eq. 3 $$s_{Boundary}(t) = \frac{Q}{4\pi T} W(u_{2a})$$

with $u_{2a} = \frac{S \cdot (2a)^2}{4Tt}$.

Usually the distance *a* between the pumping well and the boundary is large compared with the effective borehole radius *r*. At early times t, u_{2a} is large against u_r of the Theis equation. Since the well function W(u) is small at large $u=u_{2a}$, $s_{Boundary}$ does not contribute significantly to the total drawdown $s_{total}=s_{Well}+s_{Boundary}$ at early times.

- **Two barrier boundaries intersecting at 90⁰**

The case of perpendicular barrier boundaries is illustrated in Fig. 2:

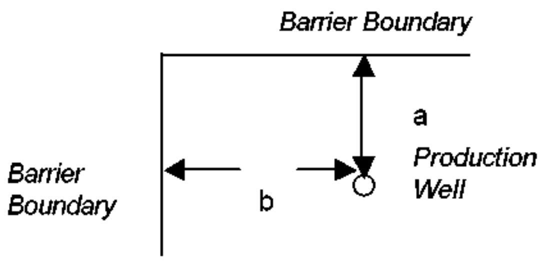


Fig. 2: Two Barrier Boundaries Intersecting at 90^0

In this case three image wells are needed to describe the drawdown in the pumping well. The drawdown due to the three image wells is expressed by the following equation:

Eq. 4 $$s_{Boundary}(t) = \frac{Q}{4\pi T}\{W(u_{2a}) + W(u_{2b}) + W(u_{2c})\}$$

with $u_{2a} = \dfrac{S \cdot (2a)^2}{4Tt}$, $u_{2b} = \dfrac{S \cdot (2b)^2}{4Tt}$, $u_{2c} = \dfrac{S \cdot (2c)^2}{4Tt}$, $2c = \sqrt{(2a)^2 + (2b)^2}$.

- **Two parallel boundaries**

In the case of two parallel boundaries (Fig. 3), an infinite number of image wells is necessary to account for the drawdown due to the boundary.

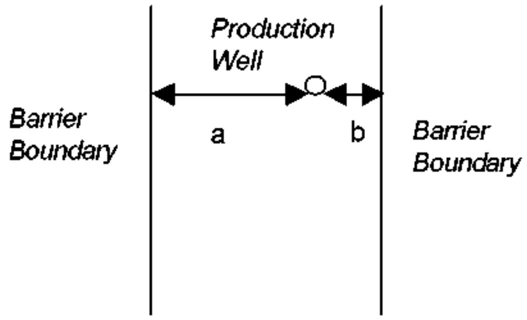


Fig. 3: Two Parallel Barrier Boundaries

The following formula approximates the influence of the boundaries by taking into account the eight closest image wells:

Eq.5

$$s_{Boundary}(t) \approx \frac{Q}{4\pi T}\{W(u_{2a}) + W(u_{2b}) + 2W(u_{2a+2b}) + W(u_{2a+4b}) + W(u_{4a+2b}) + 2W(u_{4a+4b})\}$$

with $u_{2a} = \dfrac{S \cdot (2a)^2}{4Tt}$, $u_{2b} = \dfrac{S \cdot (2b)^2}{4Tt}$, $u_{2a+2b} = \dfrac{S \cdot (2a+2b)^2}{4Tt}$, etc.

- **Closed square boundary**

A closed square aquifer is described as surrounded by barrier boundaries (Fig. 4).

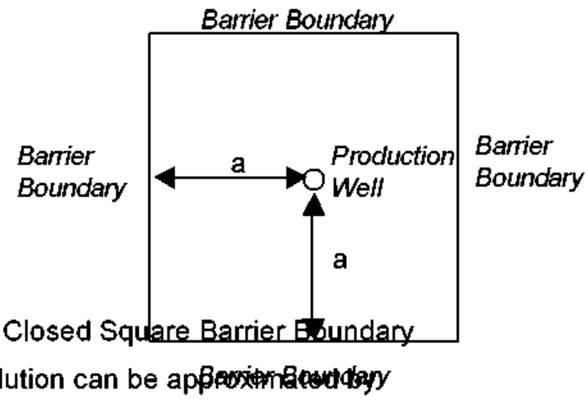


Fig. 4: Closed Square Barrier Boundary

The solution can be approximated by

Eq. 6 $\quad s_{Boundary}(t) \approx \dfrac{Q}{4\pi T}\{W(u_{2a}) + W(2u_{2a})\}$ for $\dfrac{Tt}{Sa^2} < \dfrac{1}{\pi}$

Eq. 7

$$s_{Boundary}(t) \approx \frac{Q}{4\pi T}\{-2.6084 - \frac{4}{\pi}\frac{3}{2}e^{-\frac{Tt\pi^2}{Sa^2}} + \frac{Tt\pi}{Sa^2} + 2\ln[\frac{2a}{r}]\} \text{ for } \frac{1}{\pi} \leq \frac{Tt}{Sa^2} < 1$$

and

Eq. 8 $$s_{Boundary}(t) \approx \frac{Q}{4\pi T}\{-2.6084 + \frac{Tt\pi}{Sa^2} + 2\ln[\frac{2a}{r}]\} \text{ for } \frac{Tt}{Sa^2} \geq 1$$

B4. RECHARGE CALCULATIONS: THE CHLORIDE AND EARTH METHODS *(Taken directly from Van Tonder and Xu, 2001)*

1. The Chloride Method

General Equation: $R = (P\,Cl_p + D)/Cl_w$

[R = recharge (mm/a); P = mean annual precipitation (mm/a); Cl_p = chloride in rain (mg/l); D = dry chloride deposition (mg/m^2/a); Cl_w = chloride concentration (mg/l) in soil water below active root zone in unsaturated zone OR Cl_w = chloride concentration (mg/l) of groundwater where for many boreholes the Cl_{gw} = harmonic mean of the Cl content in the boreholes].

Assumptions: The assumptions necessary for successful application are that (1) there is no source of chloride in the soil water or groundwater other than that from precipitation, (2) chloride is conservative in the system, (3) steady-state conditions are maintained with respect to long-term precipitation and chloride concentration in that precipitation, and in the case of the unsaturated zone, (4) a piston flow regime, which is defined as downward vertical diffuse flow of soil moisture, is assumed. However, this assumption may be invalidated if the flow through the unsaturated zone is along preferred pathways.

2. The EARTH Method

EARTH= Extended model for Aquifer Recharge and soil moisture Transport through the unsaturated Hardrock

General Equation: Sdh/dt=R-h/DR

[R = recharge (m^3/month); S = specific yield and dh/dt = change in water level head during one month; DR=drainage resistance (a site specific parameter); h=groundwater level]

Equation 1: Linear transfer function: $h_i = h_{i-1} - \Delta t\, h_{i-1}/DR + \Delta t\, R_i/S$

DR=$L^2/\beta T$, L=length of flow path; β=2 for radial and =4 for parallel flow; T=transmissivity

Δt=time interval (1 month)

To obtain unique fit, the value of S must be known a priori

Data Requirements

- Monthly water levels and precipitation

B5. EQUATIONS FOR MEMBERSHIP FUNCTIONS FOR THE SUSTAINABLE RISK ASSESSMENT

Membership functions for blow yield, storativity and recharge are cosine graphs in the form:

$$\text{Membership} = \left[\frac{1}{2} \times (\cos(((I - U) * \pi / \text{stretch}) - \pi) + 1)\right]$$

where

Membership	=	A value between 0 and 1
I	=	Input (the value given by the user/calculated by the DT for blow yield, storativity or recharge)
U	=	Unfavourable limit
Stretch	=	Absolute value (favourable limit – unfavourable limit)

The membership function for the pumping rate is:

$$\text{Membership} = 1 - \left[\frac{1}{2} \times (\cos(((I - F) * \pi / \text{stretch}) - \pi) + 1)\right] \text{ where}$$

I = Input (the value given by the user for pumping rate)

F = Favourable limit

The membership function for drawdown is calculated by firstly determining the power n from the following equation:

$$n = \frac{\log 0.5}{\log\left(\frac{x}{U}\right)}$$

where

$$x = 0.7U + \left(\frac{1.7(U-10)}{10}\right)$$

The membership function can now be determined as:

$$\text{Membership} = 1 - \left(\frac{I}{U}\right)^n$$

where

I = Drawdown determined by DT.

B6. IMPORTED PUMPING TEST DATA FOR BOREHOLE UO5

Pumping test data is stored in comma deliminated *.ddn file. The format of the file is:

Q (pumping rate in L/s)

Time (min), drawdown (m) or waterlevels (m) x n

where n is the number of observations

The UO5.ddn file used in the intermediate sustainability assessment:

1.25	
1.5	0.20893
...	...
...	...
...	...
380.5	2.617282
390.5	2.640574

B7. FORMAT OF IMPORTED GEOGRAPHICAL DATA FILES

Borehole data is stored in comma deliminated *.bhl file. The file format is:

Number, name, x-coordinate, y-coordinate x n

where n is the number of boreholes

The Campus test site boreholes *.bhl file is:

1	UO1	-78888.7	-21052.1
2	UO2	-78883.6	-21055.4
3	UO3	-78892.4	-21066.2
4	UO4	-78892.9	-21068.9
5	UO5	-78893.6	-21071
6	UO6	-78894.6	-21076.3
7	UO7	-78899.4	-21075
8	UO8	-78898.4	-21069.8

9	UO9	-78897.1	-21065.4
10	UO10	-78844.7	-21066.7
11	UO11	-78868.3	-21102.4
12	UO12	-78887.5	-21135.7
13	UO13	-78910.8	-21044.9
14	UO14	-78919.3	-21070.9
15	UP15	-78898.7	-21092.9
16	UP16	-78912.1	-21097.8
17	UO17	-78878	-21057.1
18	UO18	-78875.6	-21080.2
19	UO19	-78893.4	-21113.8
20	UO20	-78909.8	-21072.8
21	UO21	-78996.8	-21051.5
22	UO22	-78954.1	-21165.5
23	UO23	-78967	-21136
24	UO24	-78990	-21165

Boundary data is stored in comma deliminated *.brd file. The file format is:

Boundary type (will always be NOFLOW)

x-coordinate, y-coordinate x n (where n is the number of points given for the boundary)

END

The example discussed in Chapter 4 does not have a boundary file.

B8. FORMAT OF INPUT FILE FOR EARTH MODEL

Water level data is stored in comma deliminated *.rwl file. The file format is:

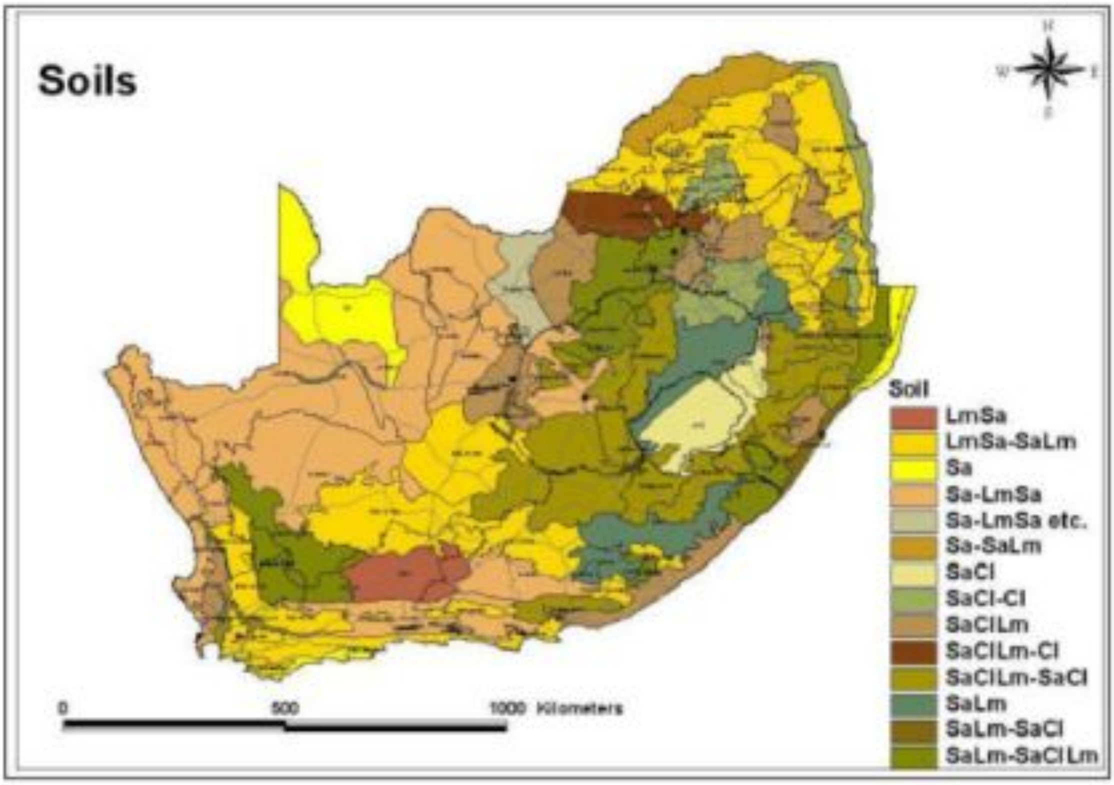
Month no, water level (m) x n

where n is the number of water level readings

APPENDIX C

Groundwater Contamination Risk Assessment Information

C1. SOILS MAP



Where

Sa = Sand

Lm = Loam

Cl = Clay

C2. WATER QUALITY GUIDELINES

Drinking water guidelines are taken from Quality of Domestic Water Supplies (DWAF, 2001). However, as the domestic water guidelines are limited, numerous other drinking water guidelines were included:

- World Health Organisation
- South African
- US Environmental Protection Agency
- Australian
- European Economic Community

These guidelines are available on the following web sites:

- www.waterquality.Cr.org.au/guide.htm
- http://www.ehl.cc/pdf/TGWD-revised.pdf

As some of the values vary, the lowest value from all the guidelines was chosen for the ideal and unacceptable limits.

Rating for the rapid assessment where there are no concentrations

Contaminant	Rapid rating
2,3,7,8,-TCDD (Dioxin)	Effects common
2,4,5,-TP (Silvex)	Long-term effects
Acephate	Few effects
Acrylamide	Effects common
Aldicarb	Effects common
Aldrin & Dieldrin	Death
Aluminium	Long-term effects
Ammonia	Few effects
Antimony	Long-term effects
Arsenic	Death
Asbestos	Long-term effects
Atrazine	Long-term effects
Barium	Few effects
Bentazon	Few effects
Benzene	Death
Benzo(a)Pyrene	Effects common
Beryllium	Effects common
Boron	Few effects
Bromacil	Few effects
Bromate	Effects common
Bromodichloromethane	Few effects
Bromoform	Few effects
Cadmium	Long-term effects
Carbaryl	Effects common
Carbofuran	Effects common

Carbon Tetrachloride	Long-term effects
Chlordane	Death
Chlordimeform	Few effects
Chlorfenvinphos	Long-term effects
Chlorine	Death
Chlorine dioxide	Effects common
Chloroform	Few effects
Chlorphyrifos	Few effects
Chromium III	Few effects
Chromium VI	Long-term effects
Cobalt	Long-term effects
Copper	Few effects
Cyanide	Death
Cyhexatin	Few effects
DDT	Effects common
Diazinon	Death
Dibromo-3-chloropropane 1,2-	Few effects
Dicamba	Few effects
Dichloroacetic Acid	Death
Dichlorobenzene 1,2-	Long-term effects
Dichlorobenzene 1,4-	Long-term effects
Dichloroethane 1,1	Few effects
Dichloroethane 1,2	few effects
Dichloroethene 1,2-(Cis)	Long-term effects
Dichloromethane	Long-term effects
Dichloropropene 1,3-	Effects common
Dichlorvos	Few effects
Dicofol	Long-term effects
Dieldrin	Death
Dinoseb	Long-term effects
Diquat	Few effects
Disulfoton	Few effects
Diuron	Few effects
Endosulfan	Death
Endrin	Death
Ethylbenzene	Long-term effects
Ethylene Dibromide (EDB)	Effects common
Fluoride	Death
Formaldehyde	Few effects
Glyphosate	Few effects
Heptaclor	Effects common
Heptaclor Epoxide	Effects common
Hexachlorobenzene	Effects common
Hexachlorobutadiene	Few effects
Hexachlorocyclopentadiene	Effects common
Lead	Effects common
Lindane	Effects common
Manganese	Few effects

Mercury	Effects common
Methidathion	Effects common
Methoxychlor	Few effects
Metolachlor	Effects common
Metribuzin	Long-term effects
Molydbenum	Few effects
Monocrotophos	Few effects
Nickel	Few effects
Nitrate	Death
Nitrate/Nitrite	Death
Nitrite	Death
PAH	Effects common
Parathion	Death
PCBs	Long-term effects
Pentachlorophenol	Effects common
Phenols	Effects common
Piperonyl Butoxide	Few effects
Posphorus	Effects common
Propoxur	Effects common
Radon	Long-term effects
Selenium	Few effects
Silver	Few effects
Styrene	Few effects
Sulfate	Few effects
Thallium	Effects common
Thiram	Few effects
Tin	Few effects
Titanium	Effects common
Toluene	Long-term effects
Toxaphene	Few effects
Trichloroacetic Acid	Few effects
Trichloroethane 1,1,1,-	Effects common
Trichlorophenol 2,4,6-	Few effects
Trifluralin	Few effects
Uranium	Few effects
Uranium^{238}	Few effects
Vinyl Chloride	Effects common
Xylene	Effects common
Zinc	Few effects

C3. DIFFUSION VALUES

Parameter	Material	Diffusion Coefficient (m^2/s)	Source
H+	Water	9.31E-09	Spitz and Moreno, 1996
Na+	Water	1.33E-09	Spitz and Moreno, 1996
K+	Water	1.96E-09	Spitz and Moreno, 1996
Rb+	Water	2.06E-09	Spitz and Moreno, 1996
Cs+	Water	2.07E-09	Spitz and Moreno, 1996
Mg2+	Water	7.05E-10	Spitz and Moreno, 1996
Ca2+	Water	7.93E-10	Spitz and Moreno, 1996
Sr2+	Water	7.94E-10	Spitz and Moreno, 1996
Ba2+	Water	8.48E-10	Spitz and Moreno, 1996
Ra2+	Water	8.89E-10	Spitz and Moreno, 1996
Mn2+	Water	6.88E-10	Spitz and Moreno, 1996
Fe2+	Water	7.19E-10	Spitz and Moreno, 1996
Cr2+	Water	5.94E-10	Spitz and Moreno, 1996
Fe3+	Water	6.07E-10	Spitz and Moreno, 1996
OH-	Water	1.57E-09	Spitz and Moreno, 1996
F-	Water	1.46E-09	Spitz and Moreno, 1996
Cl-	Water	2.03E-09	Spitz and Moreno, 1996
Cl-	Clay glacial	5.00E-10	Spitz and Moreno, 1996
Cl-	Silty clay	1.00E-09	Spitz and Moreno, 1996
Cl-	Silty clay	7.40E-10	Spitz and Moreno, 1996
Cl-	Glaciolacustrine clay	5.80E-10	Spitz and Moreno, 1996
Cl-	Varved glaciolacustrine	5.80E-10	Spitz and Moreno, 1996
Cl-	Glaciomarine clay	2.00E-10	Spitz and Moreno, 1996
Br-	Water	2.01E-09	Spitz and Moreno, 1996
HS-	Water	1.73E-09	Spitz and Moreno, 1996
HCO3-	Water	1.18E-09	Spitz and Moreno, 1996
CO32-	Water	9.55E-10	Spitz and Moreno, 1996
SO42-	Water	1.07E-09	Spitz and Moreno, 1996
Dichloromethane	Fractured clay	1.24E-09	Ross and Lu, 1999
Tricholoroethylene	Fractured clay	1.01E-09	Ross and Lu, 1999
Uranium(VI)	Granite	3.00E-14	Yamaguchi *et al.*, 1997
NaCl	Sandstone (coarse)	6.33333E-11	IGS laboratory
NaCl	Sandstone (medium)	2.22778E-11	IGS laboratory
NaCl	Sandstone (medium)	1.89444E-11	IGS laboratory
NaCl	Sandstone (fine)	9.27778E-13	IGS laboratory
NaCl	Shale (coarse)	5.22222E-11	IGS laboratory
NaCl	Shale (medium)	4.94444E-11	IGS laboratory
NaCl	Shale (fine)	2.10833E-10	IGS laboratory
NaCl	Quartzite	5E-11	IGS laboratory
Na_2SO_4	Sandstone (coarse)	5.25E-11	IGS laboratory
Na_2SO_4	Sandstone (medium)	7.44444E-12	IGS laboratory
Na_2SO_4	Sandstone (medium)	6.69444E-12	IGS laboratory
Na_2SO_4	Sandstone (fine)	6.69444E-12	IGS laboratory
Na_2SO_4	Shale (coarse)	5.19444E-12	IGS laboratory
Na_2SO_4	Shale (medium)	6.08333E-12	IGS laboratory

C4. DISPERSIVITY VALUES *(Taken directly from Spitz and Moreno, 1996)*

Material	Migration Distance (m)	Dispersivity α_L (m)
Alluvium	15	3
Alluvium	40	3
Alluvium	15500	30.5
Alluvium, derived from tuff	91	20
Alluvium (gravels)	25	1
Alluvium (gravels)	290	41
Basalt, brecciated	17	0.60
Basalt, lava, and sediments	2000	91
Basalt, lava, and sediments	20000	910
Chalk	8	1.0
Chalk, fractured	8	3.1
Crystalline rock, fractured	538	134
Dolomite, fractured	21	2.1
Dolomite, fractured	23	5.2
Dolomite, fractured	55	38.1
Dolomite, fractured limestone	122	15
Dolomite	250	7
Granite, fractured	5	0.5
Granite, fractured	17	2
Gravel, fluvioglacial	10	5
Gravel with cobbles	54	1.4
Gravel	700	200
Limestone	91	11.6
Limestone	2000	170
Limestone, fractured	490	6.7
Limestone, fractured	32000	23
Sandstone	4	0.1
Sandstone and alluvial sediments	50000	200
Sandstone with silt and clay layers	28	1.0
Sand	3	0.03
Sand	5	0.1
Sand	6	0.18
Sand	8	0.5
Sand	13	1.0
Sand	100000	20000
Sand, glaciofluvial	11	0.08
Sand, glaciofluvial	90	0.5
Sand, glaciofluvial	600	45
Sand, glaciofluvial	700	7.6
Sand, glaciofluviatile	90	0.43
Sand, glaciofluviatile	600	45
Sand, fluvial	25	1.6
Sand, fine with glacial till	4	0.06
Sand, medium, to fine	57	1.5
Sand, medium to coarse	250	0.96
Sand, medium, layered	38	4.0
Sand and gravel	2	0.015
Sand and gravel	18	0.26

Sand and gravel	25	11
Sand and gravel	150	25
Sand and gravel	43400	91.4
Sand, gravel and silt	11	2
Sand, gravel and silt	43	11
Sand, silt and gravel	16	1
Sand, silt and gravel	79	15.2
Sand, silt and clay	57	0.76
Sand and gravel, very heterogeneous	200	7.5
Sand and gravel, glaciofluvial	3500	6
Sand and gravel, glaciofluvial	20000	30.5
Sand and gravel, glaciofluviatile	4000	460
Sand and gravel with cobbles	6	11
Sand and gravel with clay lenses, alluvial	800	15
Sand and gravel with clay lenses, alluvial	1000	12
Sand and gravel, layered and silty	10	0.7
Sand and gravel, layered and silty	100	8
Sand and gravel, layered and silty	500	58
Sand and gravel with clay lenses	19	2.5

C5. EQUATIONS USED TO DETERMINE CONTAMINANT CONCENTRATIONS IN THE COMPREHENSIVE CONTAMINATION RISK ASSESSMENT *(Taken directly from Fetter, 1999)*

Continuous injection of a contaminant in a two-dimensional flow field:

$$C(x,y) = \frac{C_0(Q/b)}{2\pi(D_L D_T)^{1/2}} \exp\left(\frac{vx}{2D_L}\right) K_0\left[\left(\frac{v}{2D_L}\left(\frac{x^2}{D_L}+\frac{y^2}{D_T}\right)\right)^{1/2}\right]$$

where

C = Concentration at position (x,y)

C_0 = Initial concentration of contaminant

Q = Rate at which contaminant is being injected

b = Thickness of the aquifer over which the contaminant is being injected

D_L = Dispersion coefficient parallel to the principal flow direction

D_T = Dispersion coefficient perpendicular to the principal flow direction

K_0 = Modified Bessel function of the second kind and zero order

and

$$v = \frac{K}{n_e}\frac{dh}{dl}$$

where

K = Hydraulic conductivity

n_e = Effective porosity

$\frac{dh}{dl}$ = Hydraulic gradient

Slug injection of a contaminant in a two-dimensional flow field:

$$C(x,y,t) = \frac{C_0 A}{4\pi t (D_L D_T)^{1/2}} \exp\left[-\frac{((x-x_0)-v_x t)^2}{4D_L t} - \frac{(y-y_0)^2}{4D_T t}\right]$$

where

C(x,y,t)	=	Concentration at position (x,y) at time t
A	=	Area over which contaminant is being injected
(x_0,y_0)	=	Position at which contaminant is being injected
t	=	Time of slug injection

Mass transport equation used to calculate dispersivity values:

$$C = \frac{C_0}{2}\left[\operatorname{erfc}\left(\frac{L - vt}{2\sqrt{Dt}}\right)\right]$$

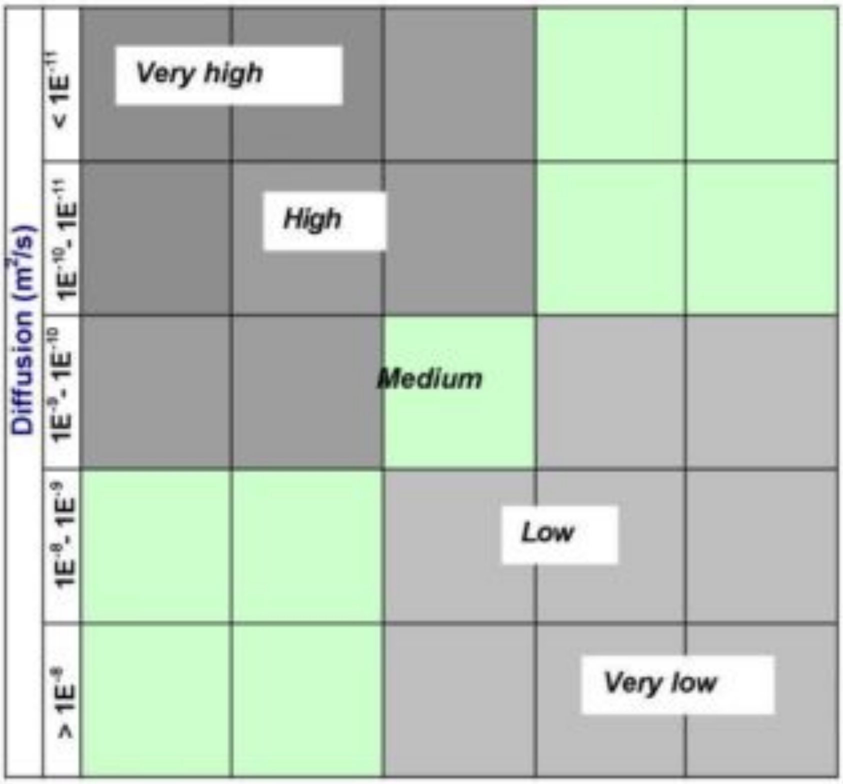
where D = αv and

α	=	Dispersivity
L	=	Distance between source and point at which concentration must be determined
t	=	Time of injection

C6. MEMBERSHIP FUNCTION RANGE FOR CONTAMINANT PROPERTIES

The membership function for the range of pollutant properties listed in Table 5.4 is dependent on diffusion (which includes matrix diffusion) and dispersion expressed in terms of dispersivity. The values are determined according to the following matrix:

	Longitudinal Dispersivity (m)				
	0 – 25	**25 – 50**	**50 – 100**	**100 – 200**	**> 200**



C7. RANGE OF HYDRAULIC CONDUCTIVITY (K) VALUES *(Taken from Freeze and Cherry, 1979)*

Rock type	K (m/d) minimum	K (m/d) maximum
Gravel	10	1.00E+05
Sand	1.00E+00	1.00E+04
Silt	1.00E-03	10
Clay	1.00E-07	1.00E-02
Sandstone	1.00E-04	1.00E+01
Limestone, dolomite	1.00E-03	1.00E+03
Karst limestone	1.00E+00	1.00E+04
Shale	1.00E-07	1.00E-02
Basalt	1.00E-05	1.00E-01
Fractured basalt	1.00E-01	1.00E+04
Dense crystalline rock	1.00E-08	1.00E-04
Fractured crystalline rock	1.00E-02	1.00E+02

C8. RANGE OF POROSITY VALUES *(Taken from Freeze and Cherry, 1979)*

Formation	Porosity Range
Gravel	0.25 - 0.4
Sand	0.25 - 0.5
Silt	0.35 - 0.5
Clay	0.4 - 0.7
Fractured basalt	0.1 - 0.5
Karst limestone	0.1 - 0.5
Sandstone	0.05 - 0.3
Limestone, dolomite	0 - 0.2
Shale	0 - 0.1
Fractured crystalline rock	0 - 0.1
Dense crystalline rock	0 - 0.05
Other	
Gabbro weathered	0.43
Granite weathered	0.455
Granite	5.00E-03
Granite, fractured	0.05

APPENDIX D

Health Risk Assessment Information

D1. CLASSIFICATION OF MICROBIOLOGICAL AGENTS FOR RAPID ASSESSMENT

(Taken from Quality of Domestic Water Supplies, 2001 and Canadian Material Safety Data Sheets, 2001)

Infectious agent	Rating
Aerococcus spp.	Effects common
Aeromonas hydrophila	Effects common
Ancylostoma duodenale	Long-term effects
Ascaris lumbricoides	Effects common
Ascaris spp.	Death
Balantidium coli	Effects common
Burkholderia (Pseudomonas) pseudomallei	Death
Campylobacter	Effects common
Citrobacter spp.	Death
Clostridium difficile	Death
Clostridium perfringens	Effects common
Clostridium tetani	Death
Clostridium spp.	Effects common
Coxsackievirus	Long-term effects
Cryptosporidium parvum	Effects common
Echinococcus granulosus	Long-term effects
Echovirus	Death
Edwardsiella tarda	Effects common
Entamoeba coli	Death
Entamoeba histolytica	Death
Enterobacter spp.	Effects common
Escherichia coli, enterohemorrhagic	Death
Escherichia coli, enteroinvasive	Effects common
Escherichia coli, enteropathogenic	Effects common
Escherichia coli, enterotoxigenic	Effects common
Fasciola hepatica, Fasciola gigantica	Effects common
Faecal coliforms	Effects common
Giardia lamblia	Effects common
Hepatitis A virus	Death
Hepatitis E virus	Death
Human rotavirus	Death
Klebsiella spp.	Effects common
Leptospira interrogans	Death
Micrococcus spp.	Few effects
Naegleria fowleri	Death
Norwalk virus	Death
Plesiomonas shigelloides	Effects common
Proteus spp.	Effects common
Pseudomonas spp.	Death
Rotavirus	Death
Salmonella choleraesuis	Death

Salmonella spp.	Effects common
Salmonella typhi	Death
Schistosoma spp.	Effects common
Serratia spp.	Death
Shigella dysenteriae	Effects common
Shigella spp.	Death
Streptobacillus moniliformis	Effects common
Taenia solium	Effects common
Total coliforms	Effects common
Vibrio cholerae, serogroup O1, serogoup O139	Death
Yersinia enterocolitica, Yersinia pseudotuberculosis	Death

D2. PARAMETERS FOR MICROBIOLOGICAL ASSESSMENT *(Taken from Rose and Gerba, 1991)*

Micro-organism	α	β	r
Campylobacter	0.039	55	-
Salmonella	0.33	139.9	-
Salmonella typhi	0.21	5531	-
Shigella	0.16	155	-
Shigella dysenteriae	0.5	100	-
Shigella flexneri 2A##	0.2	2000	-
Vibrio cholera classical	0.097	13020	-
Vibrio clolera El Tor	2.7×10^{-5}	1.33	-
Poliovirus 1	15	1000	-
Poliovirus 3	0.5	1.14	-
Echovirus 12	1.3	75	-
Rotavirus	0.232	0.247	-
Entamoeba coli	0.17	1.32	-
Entamoeba histolytica	13.3	39.7	-
Giardia lamblia	-	-	0.0199
Cryptosporidium parvum	-	-	0.00419

D3. AVERAGE VALUES USED IN THE INTERMEDIATE HEALTH RISK ASSESSMENT *(Taken directly from Genthe, 1998)*

- It is assumed that an adult weighs 70 kg.
- It is assumed that a child weighs 10 kg.
- It is assumed that a person drinks 2 liters of water a day.
- It is assumed that a person inhales 20 m^3 of air a day.
- It is assumed that the average lifetime of human is 70 years.

D4. EQUATIONS FOR MEMBERSHIP FUNCTIONS FOR THE INTERMEDIATE AND COMPREHENSIVE HEALTH RISK ASSESSMENTS

Membership functions for radiation and carcinogens are cosine graphs in the form:

$$\text{Membership} = \left[\frac{1}{2} \times (\cos(((I - U) * \pi / \text{stretch}) - \pi) + 1)\right]$$

where

Membership	=	A value between 0 and 1
I	=	Input (the value calculated by the DT radiogenic/carcinogenic risk)
U	=	Unfavourable limit
Stretch	=	Absolute value (favourable limit – unfavourable limit)

The cosine membership function for infection, toxin and size of exposed population is:

$$\text{Membership} = 1 - \left[\frac{1}{2} \times (\cos(((I - F) * \pi / \text{stretch}) - \pi) + 1)\right]$$

where

I	=	Input (the value given by the user for size of population OR the risks calculated by the DT for toxins and infection)
F	=	Favourable limit

APPENDIX E

Ecological Risk Assessment Information

E1. BREAKDOWN OF VEGETATION TYPES AND THEIR AVERAGE ROOT DEPTHS
(Taken directly from Scott and Le Maitre, 1998)

Vegetation type	Mean root depth (m)
Trees - evergreen oaks, eucalypts	12.6
Trees - conifers	3.8
Trees - overall	7
Shrubs - evergreen, mediterranean	3.5
Shrubs - overall	5.1
Grasses and other herbaceous plants	2.6
Herbaceous crop plants	2.1
Desert trees and shrubs - evergreen or deciduous	9.5
Tropical savanna/grassland vegetation	15
Temperate grassland (prairie)	2.6
Tropical forest - evergreen	7.3

E2. GROUNDWATER-SURFACE WATER INTERACTION *(Taken directly from Scott and Le Maitre, 1998)*

Interaction	South African Rivers
Influent	Kuruman River from Frylinckspan Molopo River from Tshidilamolomo
Effluent	Upper reaches of perennial rivers Vaal River Olifants River Tugela River Blyde River Komati River
Intermittent	Streams in the Karoo Salt River Kamdeboo River Sundays River Brak River
Detached	Steeply graded and dry, rocky stream beds particularly in arid north-western parts of South Africa Noseob River

E3. BASE FLOW VALUES FOR PRIMARY CATCHMENTS *(Taken directly from Vegter and Pitman, 1996)*

Drainage region	Area (km^2)	MAP (mm)	MAR (10^6m^3)	MAR (mm)	MAR (%MAP)	Baseflow (10^6m^3)	Baseflow (mm/a)	Baseflow (%MAP)	Baseflow (%MAR)
A	109610	528	2176	19.9	3.8	690	6.3	1.2	31.7
B	73550	620	2651	36	5.8	758	10.3	1.7	28.6
C	196293	571	4298	21.9	3.8	606	3.1	0.5	14.1
D	409621	315	6987	17.1	5.4	947	2.3	0.7	13.6
E	49063	212	1008	20.5	9.7	102	2.1	1	10.1
F	28623	129	24	0.8	0.6	0	0	0	0
G	25312	476	1986	78.5	16.5	250	9.9	2.1	12.6
H	15530	545	2059	132.6	24.3	245	15.8	2.9	11.9
J	45134	260	662	14.7	5.6	50	1.1	0.4	7.6
K	7220	763	1307	181	23.7	298	41.3	5.4	22.8
I	34731	283	495	14.3	5	46	1.3	0.5	9.3
M	2630	555	151	57.4	10.3	10	6.6	1.2	6.6
N	21428	330	279	13	3.9	2	0.1	0.09	0.7
P	5322	560	174	32.7	5.8	4	0.8	0.1	2.3
Q	30243	410	519	17.2	4.2	29	1	0.2	5.6
R	7936	675	580	73.1	10.8	87	11	1.6	15
S	20485	610	1043	50.9	8.3	209	10.2	1.7	20
T	46684	860	7397	158.4	18.4	1526	32.7	3.8	20.6
U	18321	935	3128	170.7	18.3	868	47.4	5.1	27.7
V	29046	829	3994	137.5	16.6	770	26.5	3.2	19.3
W	59200	825	6533	110.4	13.4	2000	33.8	4.1	30.6
X	31157	715	3361	107.9	15.1	1370	44	6.1	40.8

E4. EQUATIONS FOR MEMBERSHIP FUNCTIONS FOR THE ECOLOGICAL RISK ASSESSMENT

Membership functions for difference between groundwater level and root depth, and base flow versus abstraction, are cosine graphs in the form:

$$\text{Membership} = \left[\frac{1}{2} \times (\cos(((I - U) * \pi / \text{stretch}) - \pi) + 1)\right]$$

where

Membership = A value between 0 and 1

I = Input (the value given by the user/calculated by the DT for ratio of base flow to discharge or difference between water level and root depth)

U = Unfavourable limit

Stretch = Absolute value (favourable limit – unfavourable limit)

The membership function for toxins is:

$$\text{Membership} = 1 - \left[\frac{1}{2} \times (\cos((I * \pi / \text{stretch}) - \pi) + 1)\right]$$

where

I = Input (the concentration value calculated by the DT)

E5. RAPID AQUATIC ECOSYSTEM GUIDELINES

For the rapid assessment the following ratings will be used as there are no concentrations available:

Contaminant	Rapid rating
Acid soluble Aluminium	Long-term effects
Un-ionised Ammonia	Death
Total Arsenic	Effects common
Atrazine	Death
Total Cadmium	Long-term effects
Total residual Chlorine	Effects common
Dissolved Chromium(VI)	Few effects
Dissolved Chromium(III)	Few effects
Dissolved Copper	Effects common
Free Cyanide	Few effects
Endosulfan	Death
Dissolved Fluoride	Few effects
Dissolved Lead	Death
Dissolved Manganese	No information available
Total Mercury	Long-term effects
Phenol	Death
Total Selenium	Death
Dissolved Zinc	Long-term effects

APPENDIX F

Decision Rules for Risk Assessments

F1. DECISION RULES FOR GROUNDWATER SUSTAINABLE RISK ASSESSMENT

Rule No.	Drawdown	Blow yield	Pumping rate	Aquifer type	Effective recharge	Conclusion
1	F	F	F	F	F	0.00
2	F	F	F	F	U	0.00
3	F	F	F	U	F	0.00
4	F	F	F	U	U	0.00
5	F	F	U	F	F	0.00
6	F	F	U	F	U	0.00
7	F	F	U	U	F	0.00
8	F	F	U	U	U	0.00
9	F	U	F	F	F	0.00
10	F	U	F	F	U	0.00
11	F	U	F	U	F	0.00
12	F	U	F	U	U	0.00
13	F	U	U	F	F	0.00
14	F	U	U	F	U	0.00
15	F	U	U	U	F	0.00
16	F	U	U	U	U	0.00
17	U	F	F	F	F	1.00
18	U	F	F	F	U	1.00
19	U	F	F	U	F	1.00
20	U	F	F	U	U	1.00
21	U	F	U	F	F	1.00
22	U	F	U	F	U	1.00
23	U	F	U	U	F	1.00
24	U	F	U	U	U	1.00
25	U	U	F	F	F	1.00
26	U	U	F	F	U	1.00
27	U	U	F	U	F	1.00
28	U	U	F	U	U	1.00
29	U	U	U	F	F	1.00
30	U	U	U	F	U	1.00
31	U	U	U	U	F	1.00
32	U	U	U	U	U	1.00

F = favourable, U = unfavourable

F2. DECISION RULES FOR GROUNDWATER CONTAMINATION RISK ASSESSMENT

Vulnerability assessment decision rules

Rule No.	Depth to groundwater	Recharge	Aquifer media	Topography	Soil media	Vadose zone	Conclusion
1	F	F	F	F	F	F	0.00
2	F	F	F	F	F	U	0.25
3	F	F	F	F	U	F	0.10
4	F	F	F	F	U	U	0.35
5	F	F	F	U	F	F	0.05
6	F	F	F	U	F	U	0.30
7	F	F	F	U	U	F	0.15
8	F	F	F	U	U	U	0.40
9	F	F	U	F	F	F	0.15
10	F	F	U	F	F	U	0.40
11	F	F	U	F	U	F	0.25
12	F	F	U	F	U	U	0.50
13	F	F	U	U	F	F	0.20
14	F	F	U	U	F	U	0.45
15	F	F	U	U	U	F	0.30
16	F	F	U	U	U	U	0.55
17	F	U	F	F	F	F	0.20
18	F	U	F	F	F	U	0.45
19	F	U	F	F	U	F	0.30
20	F	U	F	F	U	U	0.55
21	F	U	F	U	F	F	0.25
22	F	U	F	U	F	U	0.50
23	F	U	F	U	U	F	0.35
24	F	U	F	U	U	U	0.60
25	F	U	U	F	F	F	0.35
26	F	U	U	F	F	U	0.60
27	F	U	U	F	U	F	0.45
28	F	U	U	F	U	U	0.70
29	F	U	U	U	F	F	0.40
30	F	U	U	U	F	U	0.65
31	F	U	U	U	U	F	0.50
32	F	U	U	U	U	U	0.75
33	U	F	F	F	F	F	0.25
34	U	F	F	F	F	U	0.50
35	U	F	F	F	U	F	0.35
36	U	F	F	F	U	U	0.60
37	U	F	F	U	F	F	0.30
38	U	F	F	U	F	U	0.55
39	U	F	F	U	U	F	0.40
40	U	F	F	U	U	U	0.65
41	U	F	U	F	F	F	0.40
42	U	F	U	F	F	U	0.65
43	U	F	U	F	U	F	0.50
44	U	F	U	F	U	U	0.75
45	U	F	U	U	F	F	0.45
46	U	F	U	U	F	U	0.70
47	U	F	U	U	U	F	0.55
48	U	F	U	U	U	U	0.80
49	U	U	F	F	F	F	0.45
50	U	U	F	F	F	U	0.70
51	U	U	F	F	U	F	0.55
52	U	U	F	F	U	U	0.80
53	U	U	F	U	F	F	0.50
54	U	U	F	U	F	U	0.75
55	U	U	F	U	U	F	0.60
56	U	U	F	U	U	U	0.85
57	U	U	U	F	F	F	0.60
58	U	U	U	F	F	U	0.85
59	U	U	U	F	U	F	0.70
60	U	U	U	F	U	U	0.95
61	U	U	U	U	F	F	0.65
62	U	U	U	U	F	U	0.90
63	U	U	U	U	U	F	0.75
64	U	U	U	U	U	U	1.00

F = favourable, U = unfavourable

Pollutant assessment decision rules

Rule No.	Pollutant	Duration	Properties	Conclusion
1	F	F	F	0.00
2	F	F	U	0.10
3	F	U	F	0.25
4	F	U	U	0.25
5	U	F	F	1.00
6	U	F	U	1.00
7	U	U	F	1.00
8	U	U	U	1.00

F = favourable, U = unfavourable

F3. DECISION RULES FOR GROUNDWATER HEALTH RISK ASSESSMENT

Rapid health risk assessment decision rules

Rule No	Toxicity	Carcinogeneity	Infection	Exposure	Population subgroup	Size of population	Conclusion
1	F	F	F	F	F	F	0.00
2	F	F	F	F	F	U	0.15
3	F	F	F	F	U	F	0.08
4	F	F	F	F	U	U	0.23
5	F	F	F	U	F	F	0.15
6	F	F	F	U	F	U	0.31
7	F	F	F	U	U	F	0.23
8	F	F	F	U	U	U	0.38
9	F	F	U	F	F	F	1.00
10	F	F	U	F	F	U	1.00
11	F	F	U	F	U	F	1.00
12	F	F	U	F	U	U	1.00
13	F	F	U	U	F	F	1.00
14	F	F	U	U	F	U	1.00
15	F	F	U	U	U	F	1.00
16	F	F	U	U	U	U	1.00
17	F	U	F	F	F	F	1.00
18	F	U	F	F	F	U	1.00
19	F	U	F	F	U	F	1.00
20	F	U	F	F	U	U	1.00
21	F	U	F	U	F	F	1.00
22	F	U	F	U	F	U	1.00
23	F	U	F	U	U	F	1.00
24	F	U	F	U	U	U	1.00
25	F	U	U	F	F	F	1.00
26	F	U	U	F	F	U	1.00
27	F	U	U	F	U	F	1.00
28	F	U	U	F	U	U	1.00
29	F	U	U	U	F	F	1.00
30	F	U	U	U	F	U	1.00
31	F	U	U	U	U	F	1.00
32	F	U	U	U	U	U	1.00
33	U	F	F	F	F	F	1.00
34	U	F	F	F	F	U	1.00
35	U	F	F	F	U	F	1.00
36	U	F	F	F	U	U	1.00
37	U	F	F	U	F	F	1.00
38	U	F	F	U	F	U	1.00
39	U	F	F	U	U	F	1.00
40	U	F	F	U	U	U	1.00
41	U	F	U	F	F	F	1.00
42	U	F	U	F	F	U	1.00
43	U	F	U	F	U	F	1.00
44	U	F	U	F	U	U	1.00
45	U	F	U	U	F	F	1.00

46	U	F	U	U	F	U	1.00
47	U	F	U	U	U	F	1.00
48	U	F	U	U	U	U	1.00
49	U	U	F	F	F	F	1.00
50	U	U	F	F	F	U	1.00
51	U	U	F	F	U	F	1.00
52	U	U	F	F	U	U	1.00
53	U	U	F	U	F	F	1.00
54	U	U	F	U	F	U	1.00
55	U	U	F	U	U	F	1.00
56	U	U	F	U	U	U	1.00
57	U	U	U	F	F	F	1.00
58	U	U	U	F	F	U	1.00
59	U	U	U	F	U	F	1.00
60	U	U	U	F	U	U	1.00
61	U	U	U	U	F	F	1.00
62	U	U	U	U	F	U	1.00
63	U	U	U	U	U	F	1.00
64	U	U	U	U	U	U	1.00

F = favourable, U = unfavourable

Intermediate and comprehensive risk assessment decision rules

Rule No	Toxin	Infection	Radiation	Carcinogen	Size of population	Conclusion
1	F	F	F	F	F	0.00
2	F	F	F	F	U	0.40
3	F	F	F	U	F	0.90
4	F	F	F	U	U	0.90
5	F	F	U	F	F	0.90
6	F	F	U	F	U	0.90
7	F	F	U	U	F	0.90
8	F	F	U	U	U	0.90
9	F	U	F	F	F	0.90
10	F	U	F	F	U	0.90
11	F	U	F	U	F	0.90
12	F	U	F	U	U	0.90
13	F	U	U	F	F	0.90
14	F	U	U	F	U	0.90
15	F	U	U	U	F	0.90
16	F	U	U	U	U	0.90
17	U	F	F	F	F	0.90
18	U	F	F	F	U	0.90
19	U	F	F	U	F	0.90
20	U	F	F	U	U	0.90
21	U	F	U	F	F	0.90
22	U	F	U	F	U	0.90
23	U	F	U	U	F	0.90
24	U	F	U	U	U	0.90
25	U	U	F	F	F	0.90
26	U	U	F	F	U	0.90
27	U	U	F	U	F	0.90
28	U	U	F	U	U	0.90
29	U	U	U	F	F	0.90
30	U	U	U	F	U	0.90
31	U	U	U	U	F	0.90
32	U	U	U	U	U	1.00

F = favourable, U = unfavourable

F4. DECISION RULES FOR GROUNDWATER ECOLOGICAL RISK ASSESSMENT

Decision rules for the quantity ecological risk assessment

Rule No.	gw-root depth	Type of dependence	Base flow	sw-gw interaction	Uniqueness of ecosytem	Conclusion
1	F	F	F	F	F	0
2	F	F	F	F	U	0.2
3	F	F	F	U	F	0.2
4	F	F	F	U	U	0.4
5	F	F	U	F	F	0.2
6	F	F	U	F	U	0.4
7	F	F	U	U	F	0.4
8	F	F	U	U	U	0.6
9	F	U	F	F	F	0.2
10	F	U	F	F	U	0.4
11	F	U	F	U	F	0.4
12	F	U	F	U	U	0.6
13	F	U	U	F	F	0.4
14	F	U	U	F	U	0.6
15	F	U	U	U	F	0.6
16	F	U	U	U	U	0.8
17	U	F	F	F	F	0.2
18	U	F	F	F	U	0.4
19	U	F	F	U	F	0.4
20	U	F	F	U	U	0.6
21	U	F	U	F	F	0.4
22	U	F	U	F	U	0.6
23	U	F	U	U	F	0.6
24	U	F	U	U	U	0.8
25	U	U	F	F	F	0.4
26	U	U	F	F	U	0.6
27	U	U	F	U	F	0.6
28	U	U	F	U	U	0.8
29	U	U	U	F	F	0.6
30	U	U	U	F	U	0.8
31	U	U	U	U	F	0.8
32	U	U	U	U	U	1

F = favourable, U = unfavourable

Decision rules for quality ecological risk assessment

Rule no.	Toxins	Duration	Conclusion
1	F	F	0.0
2	F	U	0.5
3	U	F	1
4	U	U	1

F = favourable, U = unfavourable