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Development of an assured systems management model for environmental decision-making

By

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(MSc)

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Promoter: Prof. I.J. van der Walt

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Declaration of candidate

I hereby solemnly declare that this thesis, *Development of an assured systems management model for environmental decision-making*, presents the work carried out by myself and to the best of my knowledge does not contain any materials written by another person except where due reference is made. I declare that all the sources used or quoted in this study are acknowledged in the bibliography.

Jacobus Johannes Petrus Vivier

March 2011

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PREAMBLE

The purpose of this study was to make a contribution towards decision-making in complex environmental problems, especially where data is limited and associated with a high degree of uncertainty. As a young scientist, I understood the value of science as a measuring and quantification tool and used to intuitively believe that science was *exact* and could *provide undisputable answers*.

It was in 1997, during the Safety Assessments done at the Vaalputs National Radioactive Waste Repository that my belief system was challenged. This occurred after there were numerous scientific studies done on the site that was started since the early 1980's, yet with no conclusion as to how safe the site is in terms of radioactive waste disposal. The Safety Assessment process was developed by the International Atomic Energy Agency (IAEA) to transform the scientific investigations and data into decision-making information for the purposes of radioactive waste management.

It was also during the Vaalputs investigations when I learned the value of lateral thinking. There were numerous scientists with doctorate and master's degrees that worked on the site of which I was one. One of the important requirements was to measure evaporation at the local weather station close to the repository. It was specifically important to measure evaporation as a controlling parameter in the unsaturated zone models. Evaporation was measured with an A-pan that is filled with water so that the losses can be measured. Vaalputs is a very dry place and water is scarce. The local weather station site was fenced off, but there was a problem in that the aardvark dug below the fence and drank the water in the A-pan, so that no measurements were possible. The solution from the scientists was to put the fence deeper into the ground. The aardvark did not find it hard to dig even deeper. The next solution was to put a second fence around the weather station and again the aardvark dug below it to drink the water. It was then that Mr Robbie Schoeman, a technician became aware of the problem and put a drinking water container outside the weather station fence for the aardvark and - the problem was solved at a fraction of the cost of the previous complex solutions.

I get in contact with the same thinking patterns that intuitively expect that the act of scientific investigations will provide decision-making information or even solve the problem. If the investigation provides more questions than answers, the quest is for more and more data on more detailed scales. There is a difference between problem characterization and solution identification. Problem characterization requires scientific and critical thinking, which is an important component but that has to be incorporated with the solution identification process of creative thinking towards decision-making.

I am a scientist by heart, but it was necessary to realise that apart from research, practical science must feed into a higher process, such as decision-making to be able to make a practical difference.

The process of compilation of this thesis meant a lot to me as I initially thought of doing a PhD and then it changed me, especially in the way I think. This was a life changing process, which is good. As Jesus said in Mathew 3:2 And saying, Repent (*think differently; change your mind*, regretting your sins and changing your conduct), for the kingdom of heaven is at hand.

Development of an assured systems management model for environmental decision-making

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ABSTRACT

Decision-making is one of the most important aspects in life. It involves choosing from a number of options available or developing new options. The aim of this research was to develop a decision-making methodology that can be used in complex environmental problems where data is limited and associated with a high degree of uncertainty. Decision-making is scaled from a strategic or planning to a tactical level that takes the detail and sequences into account. Environmental decision-making should adhere to the principle of sustainability where science serves as a quantification tool. The scientific approach is reductionist and the environmental management approach is holistic in nature. Decision-making is an integration of science for measurement and management to get people to plan and act.

In the decision-making process, data is analysed to become information which when interpreted becomes knowledge that is used as the basis for decisions. An accumulation and arrangement of data provides information. It was found that the decision-making process follows a logarithmic trend that is similar to the law of diminishing returns in economics. The idealised perfect information is a goal that would enable the analyst to make perfect decisions. In the absence of perfect information, assumptions have to be made. Assumptions are not only necessary, but useful if made in the correct context. More data is not necessarily better as an optimum point is reached where it becomes more expensive to collect data than the information it provides.

The use of a systems philosophy in environmental decision-making enables the analyst to make decisions in complex problems. The *assured systems methodology for environmental decision-making* was developed based on a complex environmental water management problem. It can be used for reaching decisions in an iterative approach, even with limited data associated with a high degree of uncertainty. The methodology is based on the requirements of sustainability and makes use of the principles of precautionary approach and minimax as decision rules to limit the potential effects of uncertainty.

Key words:, Environmental decision-making, sustainability, management, groundwater sparse data, systems thinking, systems model, assured systems method, precautionary principle, minimax.

Ontwikkeling van ‘n versekerde stelsel besluitnemingsmodel vir besluitneming in omgewingsbestuur

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OPSOMMING

Besluitneming is een van die belangrikste aspekte van die alledaagse lewe. Dit behels om ‘n keuse te maak tussen ‘n aantal beskikbare opsies. Die doel van hierdie navorsing was om ‘n besluitnemingsmetodologie te ontwikkel wat gebruik kan word in komplekse omgewingsprobleme waar die data beperk is en geassosieerd is met ‘n hoë mate van onsekerheid. Besluitneming word eers gedoen op ‘n strategiese vlak vir beplanningsdoeleindes en dan op ‘n taktiese vlak waar die detail en opeenvolgings van opsies in ag geneem word. Omgewingsbesluitneming moet aan die beginsels van volhoubaarheid voldoen. Die rol van die wetenskap (wat reduksionêr is) dien as ‘n kwantifiseringshulpmiddel vir omgewingsbesluitneming wat holisties van aard is. Besluitneming is ‘n integrasie van wetenskap wat nodig is vir kwantifisering en bestuur wat nodig is om mense te mobiliseer om te beplan en te implementeer.

In die besluitnemingsproses, word data wat geanaliseer word omgeskakel na inligting wat weer omgeskakel word na kennis waarop besluite gebaseer word. Inligting bestaan uit ‘n akkumulasie en rangskikking van data. Dit is bevind dat die besluitnemingsproses ‘n logaritmiese kromme volg wat soortgelyk is aan die wet van afnemende opbrengs in ekonomie. Die geïdealiseerde perfekte inligting is ‘n doelwit wat die analis in staat sou stel om die perfekte besluit te neem. In die afwesigheid van perfekte inligting moet aannames gemaak word. Aannames is nie net nodig nie, maar ook bruikbaar, indien dit in die regte konteks gemaak word. Meer data is nie noodwendig beter nie, aangesien dit ‘n optimale punt in terme van inligting bereik waarna dit minder koste-effektief is om data in te samel as die inligting wat dit verskaf.

Die gebruik van ‘n stelselbenadering in omgewingsbesluitneming stel die analis in staat om besluite te neem aangaande komplekse omgewingsprobleme. Die versekerde stelsel metodologie vir omgewingsbesluitneming is ontwikkel as deel van hierdie studie wat gebaseer was op ‘n komplekse omgewings-waterbestuurs probleem. Dit kan gebruik word om by omgewingsbesluite uit te kom met ‘n iteratiewe benadering, selfs al is data beperk en geassosieer met ‘n hoë mate van onsekerheid. Die metodologie is gebaseer op die beperkings, van volhoubaarheid en maak gebruik van die omsigtigheids en minimax beginsels wat as ‘n besluitnemingsreël toegepas word om die effek van onsekerheid te beperk.

Kernwoorde: Omgewingsbesluitneming, volhoubaarheid, bestuur, grondwater, beperkte data, sisteem filosofie, sisteem model, versekerings stelsel metode, omsigtigheidsbeginsel, minimax.

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List of abbreviations

Abbreviation	Meaning
AMD	Acid Mine Drainage
BPEO	best practical environmental option
C	Capacity
CBD	Central Business District
CBL	Critical Base Level
CDM	Clean Development Mechanism
CFC	Chlorofluorocarbon
CGS	Council for GeoScience
CPI	Consumer Price Index
CRB	Central Rand Basin
CS	Cost per Section
CU	Cost per Unit
d	day
DDD	Diaphonic Definition of Data
DMR	Department of Mineral Resources
DWA	Department of Water Affairs
ECA	Environmental Conservation Act
EMP	Environmental Management Plan
F	Safety Factor
FERB	Far East Rand Basin
EV	Equivalent Value
GDI	General Definition of Information
GIS	Geographic Information System
RDM	Resource Directed Measures
GRDM	Groundwater Component of the Reserve
IAEA	International Atomic Energy Agency
IAM	Impact Assessment Matrix
IG	Information Gained
IT	Information Technology
ℓ	Liter
LD	Load
m	meter

Abbreviation	Meaning
M	Mega
MAE	Mean annual evaporation
MAP	Mean annual precipitation
MHI	Major Hazardous Installtion
NEMA	National Environmental Management Act
NGDB	National Groundwater Database
NPP	Nuclear Power Plant
NPC	Net Present Cost
NPV	Net Present Value
NWA	National Water Act (Act 36 of 1998)
O&M	Operation and Maintenance
OPEC	Organization of Petroleum-Exporting Countries
P	Probability
PDF	Probability Density Function
Pf	Probability of Failure
PFD	Process Flow Diagram
PFM	Process Flow Model
PIC	Pilanesberg Intrusion Complex
PPP	People Planet Profit
PV	Photovoltaic
R	Rand
Rl	Reliability
RoD	Record of Decision
s	second
SEA	Strategic Environmental Assessment
SM	Safety Margin
STP	Sewage Treatment Plant
TBL	Triple Bottom Line
TDF	Tailings Disposal Facility
TDS	Total dissolved solids
TEV	Total Economic Value
URV	Unit Reference Value
WM	With Mitigation Measures
WoM	Without Mitigation Measures
WRD	Waste Rock Dump

Abbreviation	Meaning
WUC	Western Utilities Corporation
Y2K	Year 2000 Compliant

CHAPTER 1

"We will keep making the same mistakes if we think in the same way than we did when making them"
A. Einstein

1 INTRODUCTION

Complex environmental water management problems were created in South Africa, especially in the derelict gold mining and other industries (Mining Weekly, 2010a; Handley, 2004). These problems escalated to the level that decisive action should be taken, which poses a challenge for decision-makers. Environmental water management investigations are done based on scientific methods, which in most cases identify inadequacies in data and information that highlights the uncertainties associated with these problems. The result is that additional data is usually required to scale up the scientific investigations. This leads to a requirement of more investigations where even more data gaps or complexities of the problem are identified. In these cases, where an assumed data deficiency is perceived by analysts, management objectives cannot be met as present methods of assessment requires increasing detailed data sets that are seldom available, rather than supporting the decision-making process. This leads to a divergent process that highlights uncertainty and counteracts the decision-making process.

The methods of quantification and decision-making for management purposes need to be adapted for these regional scale environmental problems. In practise, analysts aim to collect field data before performing a data sufficiency analysis and often get a false assurance by collecting inappropriate data sets or by over-collecting field data. Based on this, the following problem statement can be formulated for this study:

"A decision-making process, utilising limited and sparse data sets, is required to allow for effective environmental decision-making and sustainable resource management".

1.1 Critical questions and comments formulated during investigations into complex environmental water problems.

The following are selected critical questions and comments that were encountered in the study of a number of complex environmental water management problems. It assisted in formulating this study:

1. You do not have enough data.
2. You do not have sufficient data.
3. Where are *all* the boreholes?
4. You need more water level data.
5. You need more geological data.
6. You need more pump tests.
7. Do you have monitoring data?
8. You need monitoring data.
9. You have a non-exact solution.
10. There are not enough data for a groundwater model.
11. You have too many assumptions.
12. Your assumptions are idealistic.
13. Due to too many assumptions, the model cannot be developed.
14. How accurate is your model?
15. The geology is so complex that a model cannot be developed for it.
16. The conceptual model is a simplification of the actual geology.
17. Do you know all the parameters?
18. What if there are some parameters that you do not know?
19. How much is the recharge?
20. How do you know if the wetlands are groundwater supported?

1.2 Objectives

To investigate the problem statement, it is necessary to answer the following research questions:

1. What is decision-making and how does it relate to sustainability?
2. Is there a relationship between data and information in the decision-making process?
3. Is more data better and when is information sufficient for decision-making?
4. If groundwater is used as an example of an environmental component, what are the characteristics of these parameters and how does it influence decision-making?
5. Can a systems approach be used for characterization, understanding and decision-making in complex environmental water management problems?
6. Are models accurate enough and does it add value to the decision-making process?
7. Is it possible to apply a systems, modelling approach on a case study that is based on a complex environmental management problem?
8. Is it possible to develop an assured decision-making methodology that can be used even if data is sparse and associated with a high degree of uncertainty?

1.3 Methodology

In order to answer the research questions, the following research methodologies have been applied:

1. A literature review on the theory of decision-making in the context of sustainability.
2. Research and evaluate the relationship of data and information in the decision-making process.
3. Based on the data of three field studies, determine whether there is a trend between information and the number of data points, which should be used to map the decision-making process.
4. Identify the main parameters that are used in groundwater and based on the uncertainties associated with each parameter, critically evaluate whether it can be used in the decision-making process.
5. Research the characteristics of systems thinking and systems models to determine if it can be

used for the characterization, understanding and decision-making in complex environmental water management problems.

6. Evaluate the purpose of models to determine if and how it can be used in the decision-making process.
7. Apply a systems modelling approach on a case study of the Far East Rand Basin environmental water management problem.
8. Based on the case study, develop an assured systems management model for decision-making in complex environmental water management problems.

1.4 Thesis layout

The layout of this thesis is as follows:

1. Chapter 1: Introduction and objectives.
2. Chapter 2: Literature Review: Decision-making theory in relation to environmental management and sustainability.
3. Chapter 3: Data and information in the decision-making process.
4. Chapter 4: The characteristics of groundwater data and implication on decision-making in environmental reserve determinations.
5. Chapter 5: A systems modelling approach to environmental decision-making.
6. Chapter 6: Application of a systems approach on environmental decision-making at the Far East Rand Basin (FERB) mine water flooding problem.
7. Chapter 7: Development of an assured method for decision-making in complex environmental water management problems.
8. Chapter 8: Conclusions and Recommendations
9. References
10. Annexure A: Application of the assured systems method on the Middelburg groundwater supply problem.

CHAPTER 2

"If you fail to plan you plan to fail" Unknown

2 LITERATURE REVIEW: DECISION-MAKING THEORY IN RELATION TO ENVIRONMENTAL MANAGEMENT AND SUSTAINABILITY

2.1 Introduction

Decision-making is one of the most important characteristics of modern society. With technological advances in the information age, it has become more important and often more difficult to be able to interpret data and information to come to meaningful conclusions. The information revolution is followed by the environmental revolution (Henderson, 2010). Where everyone was talking about information technology (IT) and the Y2K problems in the late 1990's, today it is about environmental aspects of global warming, greenhouse gasses, sustainable energy and sustainable development. In the late 1990's the IT industry was in agreement that the Y2K problem could or would be a reality. The environmental community on the other hand is split in two on the issue of global warming. There are the proponents that show data to prove that global warming is a reality (Gore, 2006; Meinshausen et al, 2009) and those that argue against it (Kline, 2007; Goodwin, 2009; Lemonick, 2010). It would be expected that scientific data of temperature, carbon dioxide and other parameters measured over hundreds of years would provide a concrete conclusion in matters like these. The question is how individuals, communities and governments make decisions in these circumstances?

In this chapter, the theoretical basis of decision-making is investigated. Background is provided on the earliest known historical uses in warfare and business management where decision-making theory was developed. The basis of decision-making in environmental management must be based on sustainability, which includes economics (business management), socio-political, scientific (technical) and legal (regulatory) components or constraints (Vivier, 2006). Environmental decision-making based on the sustainability principle interacts with the economic or business, social and legal components. Decision-making in these fields are investigated to provide background information and to determine possible gaps in the process as environmental decision-making (or decision-making for sustainability) needs to account for these various disciplines.

The aim of this chapter is to determine the background philosophy,[1, 2] principles and theory of decision-making. The purpose is to lay a broad foundation as a reference and to determine how it can be used to enhance decision-making for sustainability.

2.2 What is decision-making?

Decision has the same meaning as making a choice or a judgement or to come to a conclusion[3]. A decision cannot be made unless there are options available. If a decision is not made in e.g. a case where there is only one (difficult) option to choose from it is defined as indecision. Indecision is an important cause of problems in business and the environment. Indecision by government to enforce non-compliance on mine water discharge and decant problems e.g. lead to a build up of acid mine drainage and contaminants such as radio nuclides in the Mooi River (Mining Weekly, 2010a; Winde and Stoch, 2010).

2.3 Decision-making theory

Decision-making theory is the study of making the best decision according to what is analysed in terms of gains and losses (Oxford Advanced Learners Dictionary, 2006). It is a process whereby the decision maker must choose an option (O) or sequence of options ($O_{1,2,n}$) from a given number of possible options (O_n) (Figure 2-1) (Edwards and Newman, 2000). The theory of decision-making on a strategic level originated in warfare (Von Klausewitz, 1832). It was later applied in business management (Ries and Trout, 1986) and today is widely used in technical management, engineering management and environmental management. Decision-making in the

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[1] Philosophy is a particular set or system of beliefs resulting in the search of knowledge. It includes the study of nature and meaning of the universe (Oxford English Dictionary, 2006). Love and pursuit of wisdom by intellectual means and moral self-discipline. The investigation of causes and laws underlying reality (http://www.answers.com/topic/philosophy). The meaning of the word philosophy from Greek philosophi, from philosophos, which means "lover of wisdom". (http:// www.thefreedictionary.com /philosophy).

[2] Wisdom is defined as the ability to make sensible decisions and give good advice because of the experience and knowledge that you have (Oxford English Dictionary, 2006). The ability or the result of an ability to think and act utilizing knowledge, experience, understanding, common sense and insight (Collins English Dictionary, 2006).

[3] The cognitive process of reaching a decision. A position or opinion or judgment reached after consideration. Choosing between alternative courses of action using cognitive processes - memory, thinking, evaluation, etc . The process of mapping the likely consequences of decisions, working out the importance of individual factors, and choosing the best course of action to take (www.decision-making-confidence.com/definition-of-decision-making.html). Choice made between alternative courses of action in a situation of uncertainty. Although too much uncertainty is undesirable, manageable uncertainty provides the freedom to make creative decisions (www.businessdictionary.com/definition/decision.html).

business environment is defined as a definition of problems, followed by gathering of information, then the generation of alternatives and making a choice on a course of action.

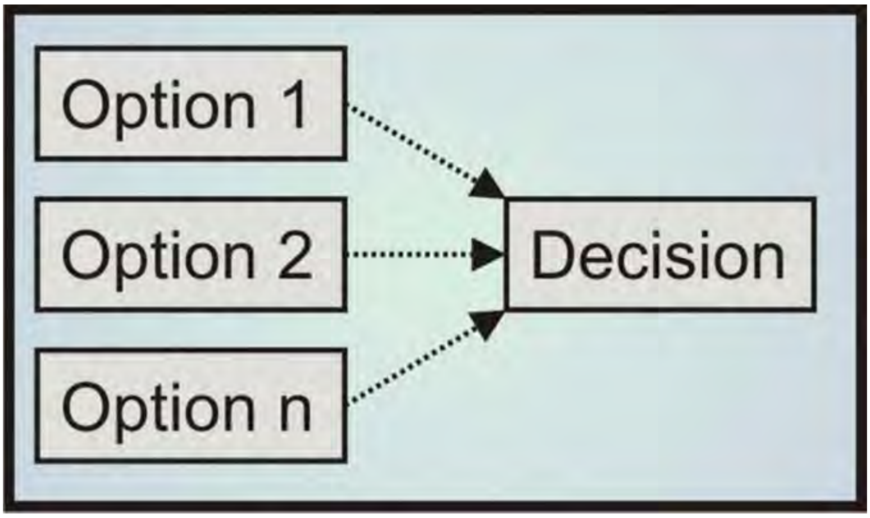

Figure 2-1 Schematic representation of a decision with alternative options.

The conditions under which decisions are made are defined as certainty, and risk or uncertainty. Decisions can either be made objectively based on *exact* information (facts and numbers) or subjectively based on personal judgement (Hellrieger, et al. 2002). Decision-making varies in terms of uncertainty. Routine decisions are typically made based on known problems with known solutions. Adaptive decisions are made where there are less well known problem types and solutions with an increase in risk. Some decisions are required under conditions of uncertainty where new problems or challenges arise for which no existing solutions exist. Under these conditions, uncertainty is high and solutions untried or even non-existing or the decision maker must make innovative decisions by finding new solutions (Figure 2-2).

There are various levels that are important in decision-making. Decisions can be made based on gut feeling, a hunch, using existing cognitive information or by following a decision-making process (Hellrieger, et al. 2002). High level decision-making is known as strategic decision-making, which is followed by more detailed tactical decision-making that is refined further by real time or operational decision-making. These levels of decision-making are investigated in more detail in the following sections.

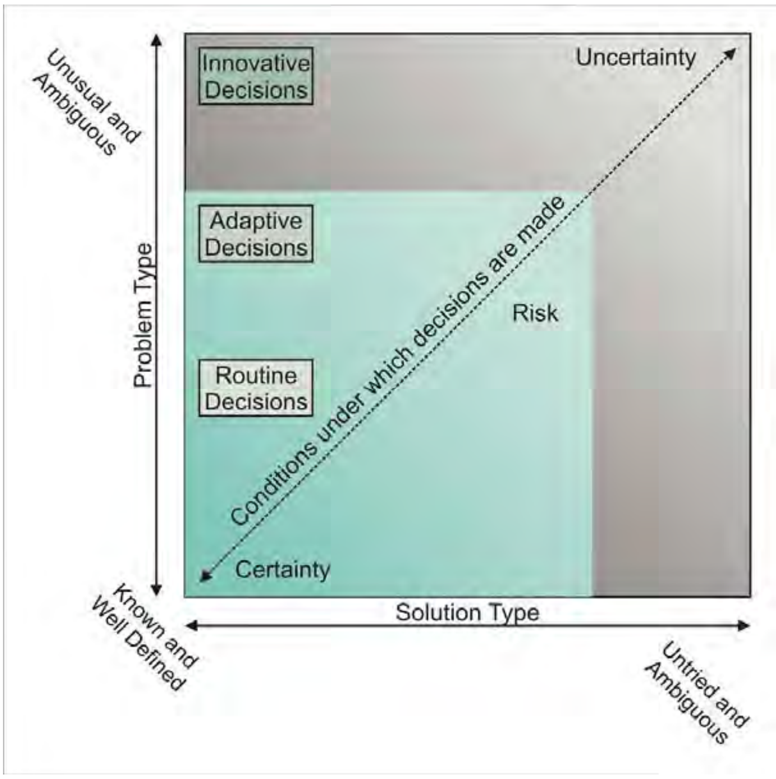

Figure 2-2 Framework for decision-making (after Hellriegel, et al. 2002).

2.3.1 Strategic decision-making

The planning or high level phase of decision-making is known as strategic[4] decision-making. It usually takes time and involves choosing a general direction in terms of the *what* without the consideration of detail (the where, when and how). Strategic planning in a business environment would be a process of diagnosing an organization's external and internal environments and the development of overall goals. In unusual cases, strategic decisions could be required in high velocity environments where a window of opportunity exists for a small period of time (Eisenhardt, 1989). Strategic planning would also include contingency planning (Hellrieger, et al. 2002). A chess player would take a strategic approach at the start of the game to determine what type of game is to be played (Linhares, 2009). A strategic assessment would be done over the whole country to select the most suitable locations on a high level e.g. the siting of nuclear power plants.

[4] Strategy is derived from the Greek word *strategos* which means "the art of the general" as it originated from warfare planning. The English meaning relates to a plan that is intended to achieve a particular purpose or to gain an advantage (Oxford English Dictionary, 2006) or the art or science of the planning or conduct of a war (Collins English Dictionary, 2006).

On a strategic level, the decision-maker zooms out of the problem detail and observes the forest whereas detailed evaluations or decisions would be to analyse individual trees. It is not possible to focus on both these ends at the same time.

2.3.2 Tactical decision-making

Tactical decision-making evaluates the detail and analysis of information with sequences to reach the goals of the overarching strategy. It considers aspects that involve questions on *how*, *where*, *when* and *who* (Hellrieger, et al. 2002). Tactical decision-making is characterised by rapidly unfolding events, multiple plausible hypotheses, high information ambiguity and sometimes severe consequence of errors (Cannon-Bowers et al. 1992; Figure 2-2). The sequence of events is an important aspect of tactical decision-making that requires critical thinking skills in short timing (Cohen et al. 1998). Fast analysis and quantification of information is an important part of tactical planning that could involve re-planning during the process.

2.3.3 Innovative decision-making

Innovative decision-making is when a decision is made without the available information or options on hand. The decision maker has to look beyond the existing data, information, options and/or circumstances and develop unique and creative solutions (Figure 2-2). Innovative decisions are characterised with a difference in usual practices and is often not done in a logical systematic way. It is associated with a high degree of uncertainty and risk (Hellrieger, et al. 2002). An example of innovative decision-making in the technical environment is when no existing methods are suitable for a task and the analyst has to develop a new method.

2.4 Decision-making traps and errors

Research has shown that most decision-makers make the same basic errors. These basic errors are (Russo and Schoemaker, 1989):

1. **Plunging In**: Starting to obtain information and make decisions without strategic planning. Delving into the detail before the direction of the decision is determined.

2. **Frame[5] blindness**: Solving the wrong problem due to the creation of a mental framework for the decision and losing sight of the most important objectives. This could occur because the decision-maker is trained in a specific discipline and would naturally consider solutions in this narrow range.

3. **Lack of Frame Control**: Lack of defining the problem in more than one way before embarking on a decision-making process. If a problem cannot be bounded or framed, it cannot be solved.

4. **Overconfidence in Personal Judgement and Intuition**: Failure to collect the most important factual information and overestimating personal assumptions and opinions, *rules of thumb* or convenient facts. This error is associated with a lack in following a formal decision-making process and relying on intuition.

5. **Shooting From the Hip**: A belief that all the information that is discovered can be kept in one's head which leads to the non-adherence to following a systematic decision-making process.

6. **Group Failure**: Making the assumption that with a group of educated (or smart) people, good or optimal choices will automatically be made. This leads to a failure due to blindly following the group's decision-making process.

7. **Limited Feedback**: A failure to objectively analyse evidence from past outcomes of processes in terms of what the real meaning is or by focusing on selective feedback information.

8. **Not Keeping Track**: Making an assumption that experience will make its lessons automatically available and failing to keep a systematic data base of results that failed in the past.

9. **Failure to Audit the Decision-making Process**: Not creating an organised approach to review and understand the decision-making process.

[5] A decision frame sets the boundaries within which a decision should be taken.

Bad decisions mostly originate from personal biases originating from mental or paradigm flaws (Hammond et al. 2003). The decision-making errors above have a golden threat in that personal approaches are followed or loose assumptions made rather than having a formal system in place that guides the decision-making process. A formal, but optimal decision-making process based on a quality system with monitoring and feedback would prevent or minimise the potential for these mistakes.

2.5 Decision-making and sustainability

Decision-making in environmental management must be based on the principle of sustainability that is also known as the triple bottom line (represented by PPP for People, Planet and Profit). Sustainable development was first described by the Brundtland Commission in 1987 as;

"development that meets the needs of the present without compromising the ability of future generations to meet their own needs" (Brundtland, 1987).

The concept of sustainability is represented by the triple bottom line representation of environment, technology (development) and social components (Figure 2-3; Gibson, 2001; Vivier, 2006). It is important to note that any decision that is made based on the sustainability principle has to conform to the three very different spheres of environmental management, technological development and social development. Social development contains three important sub-components that are important in environmental decision-making namely; political, economical, legal and management components (Figure 2-3).

Sustainability relates to an equilibrium environment where a system is in balance and time does not affect its outcome. The oil resources of the world are e.g. not a sustainable resource as it is limited in extent while the usage grows every year. The bushmen culture of Southern Africa was sustainable as the growth in population was determined or limited by the environment and a balance was naturally maintained.

Principles relevant to the achievement of sustainable development are given effect in most of South Africa's legislation and policies that include the principles of *polluter pays*, *cradle-to-grave*, *precautionary approach*, *waste avoidance* and *minimisation*, *best practicable environmental option (BPEO)* and *as low as reasonably possible (ALARP)*, (NEMA, Act 107 of 1998).

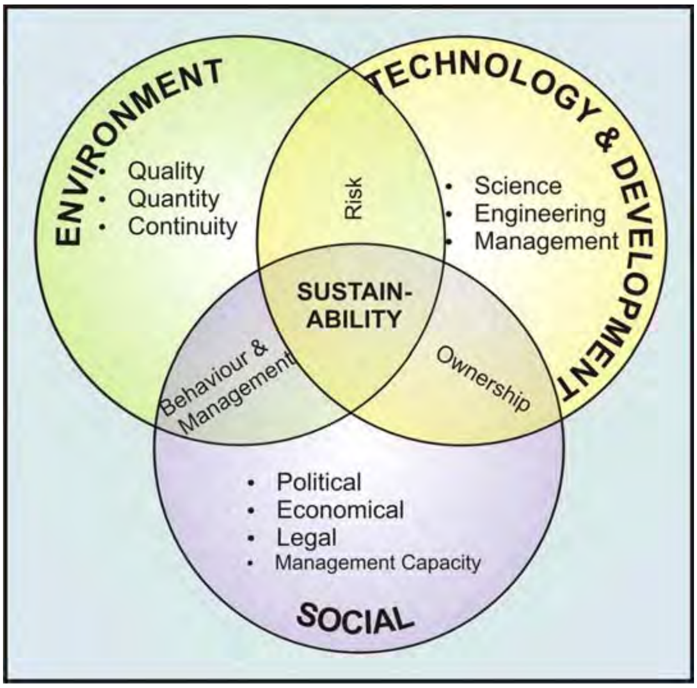

Figure 2-3 The three components of sustainability (modified after Vivier, 2006).

In practice, it is often found that decisions are made based on one component only, such as social or technological (development) (George, 2001; Fuller, 2002). This is due to discipline-specific biases that are created when only one component is considered in the decision-making process (Section 2.4). The effect of this is that an invisible pressure against the integration of the triple bottom line is generated. This invisible but opposite pressure is due to the very different nature of the three components of sustainability that has to be integrated (Figure 2-4). The effect of this pressure is an increase in risk that eventually leads to an unsustainable situation. An example of these biases in the technical sphere is where an engineer would design the most practical and least costly waste water containment dam without recognising the potential impact on groundwater or where onsite sanitation is done at the most basic level that contaminates the groundwater resource (Vivier, 2006). In the environmental-social sphere, problems with biases are encountered when environmental activists (so called *tree huggers*) that do not want any development solutions or accept that any development has an impact. In these cases, the activism leads to a non-solution to any development. The aversion to nuclear power in the 1990's and the realization of some environmental benefits, leading to the current favouritism towards it by green activists, is a good example of the problems that have been created in the past from this world view.

The purpose of the following sections is to evaluate the decision-making philosophies, principles and methodologies in the social (business or economics, legal and political), technical & development (science and engineering) and environmental spheres. The social component is more subjective and complex than the other fields and is expanded into the sub-fields of business or economics, legal and political. This is due to the fact that the environmental field is strongly influenced by politics and regulated by legal processes.

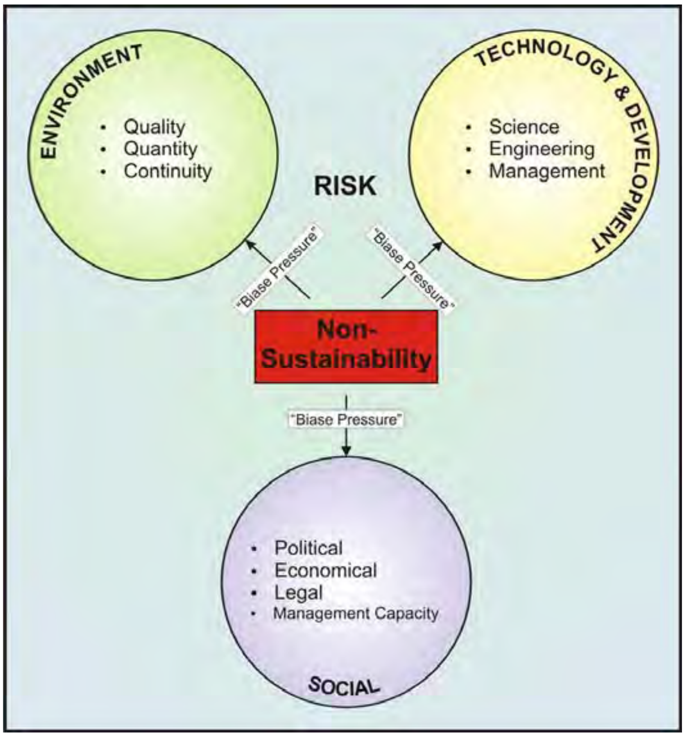

Figure 2-4 Invisible but opposite pressure factors causing non-sustainability (Figure 2-3).

2.6 Decision-making in business management (economics)

Historically, decision-making theory evolved from the military to the business environment (Ries and Trout, 1986). The decision processes are well researched and developed in aspects like general business and organizational management. There are various decision-making models that were developed for business management. These range from quantitative mathematical models based on numbers, to more subjective, innovative and political methods (Hellrieger, et al. 2002). Economics are important because it is the driver of human development and influence the environment. Poverty is one of the biggest threats to sustainability (Tyler Miller, 2005). The mitigation measures related to environmental impacts all have cost implications. For example, if there are sufficient funds available from a development, proper rehabilitation can be done to minimise environmental impacts. It is often low income operations that have high environmental impacts, such as low cost sanitation (Vivier, 2006). In this section, the decision-making methods in business management and economics are investigated.

2.6.1 The rational method

The rational method consists of a formal set of detailed actions or phases that are followed to increase the probability of making optimal decisions. The following seven phases are included in the rational model for decision-making (Hellrieger, et al. 2002; www.scribd.com):

1. Define decision-making objectives and diagnose the problem.
2. Setting of goals and structuring of objectives.
3. Identification of alternative solutions or options.
4. Compare and evaluate alternative solutions.
5. Choose among alternative solutions.
6. Implement the selected solution.
7. Follow up and control.

The aim of the rational model for decision-making is to determine the *maximum* achievement (or *best* alternative) towards the goal within the limitations of the situation. It is a detailed process that is often time consuming. The rational model for decision-making is exhaustive and in most cases too exhaustive in terms of time and cost (Simon, 1978; Eisenhardt and Zbaracki, 1992).

The rational method would e.g. be used to determine the requirement and location of a new nuclear power plant, but not for the development of a new golf course. The risk of following the rational model for certain decisions is that it could take more resources in terms of time and cost than time would permit and the window of opportunity could be lost once the whole process is followed.

2.6.2 The bounded rationality model

The bounded rationality model represents the decision-maker's inclination to select less than the best goal or alternative. It is characterised by a limited search for alternative solutions. It is accepted that information and control over external and internal environmental aspects influencing decisions will be inadequate and the *near-optimal* rather than the *best* alternatives are selected (Eisenhardt and Zbaracki, 1992; Hellrieger, et al. 2002). When one sets out to buy a television set, it is e.g. not possible to evaluate *all* the options with pros and cons that are available. This could take more time if translated to cost than the cost of the television set.

The concept of *satisficing* is associated with the bounded rationality model, where it is practical to select an acceptable goal or alternative rather than to spend time and money on an elaborate research for the best or optimal alternative (Schwartz et al. 2002). This model is suitable for decision-making using sparse data and is characterised by *limited search*. Limited search is where a detailed search is not done, the tendency is to consider options until an adequate one is found. It does not aim to consider all possible options. The general limitations associated with time and money favours the limited search option. It was found that most decision-makers satisfice rather than optimize (Schwartz et al. 2002; March, 2003).

Research indicates that adding information does not necessarily provide improved decisions. It was shown that entrepreneurs starting a new venture do not usually follow a methodical approach (Hellrieger, et al. 2002). If they did and if they would have evaluated all the information and weighted all the risks, chances are that the opportunity would pass by the time they are in a position to make a decision (Goldrat, 1994). They follow a bounded rationality rather than a rational approach.

The bounding of decisions is known as *framing*, which is a very effective way of reducing the decision domain (Section 2.4; Tversky and Kahneman, 1986; Russo and Schoemaker, 1989). A *theory of constraints* was introduced in business management that showed that business processes have a constraint/s that will determine the rate of output. In linear systems, a chain is only as strong as its weakest link. The aim is to identify and manage constraint/s in the business process rather than to aim and manage all the components (Goldratt, 1990). The same principle is applicable in the decision-making process for environmental purposes.

The bounded rationality model can be described as a *near-optimal* decision-making model as it accounts for time, cost and the adequateness of the decision (Gigerenzer and Goldstein, 1996; Conlisk, 1996). Research has shown that people are boundedly rational and that power wins battles of choice and that chance affects the course of strategic decision-making (Eisenhardt and Zbaracki, 1992).

A problem that may arise with the bounded rationality method, is the under evaluation in the decision-making process that could lead to ignorance. In cases where decisions with severe consequences have to be made, the bounded rational method could or should be developed into the rational method. The success of using the bounded rational method can be improved by specifically introducing an alternative viewpoint or auditing process in a decision process (Eisenhardt and Zbaracki, 1992).

Decision-making can be viewed as a continuum bounded by two extremes. On the one side, quick, personal judgements are made with information at hand, which is usually inadequate (i.e. the bounded rationality model with satisficing). On the other side is the rational method where a detailed decision-making process is followed (Figure 2-5). There is an increase in time and cost towards the rational method with a consequent decrease in risk. The bounded rationality method is considered as a practical (near-optimal) method (Lee and Cummins, 2004).

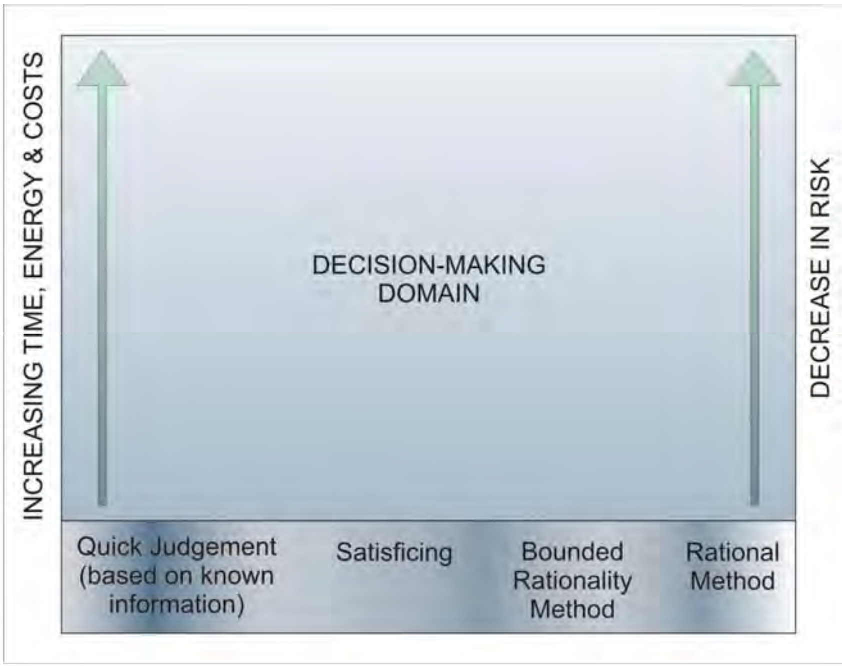

Figure 2-5 The decision-making continuum.

2.6.2.1 The analytical hierarchy process (AHP)

The AHP process is defined by collecting information on options from experts that are weighted and ranked in a matrix form. It aids in the decision-making process in that the option with the highest or lowest score is selected. It assumes that not all information can be and will be gathered and that too much information can become irrelevant. The aim is to make a decision in an organised way to ensure all relevant aspects as covered as it is based on a formal process that is followed (Saaty, 2008). The decision-making process is differentiated into a number of defined steps as follows:

1. Define the problem and determine the kind of information required.
2. Structuring of the decision hierarchy from the top down, with the goal of the decision, then the objectives from a broad perspective, followed by the intermediate to the lowest levels.
3. Construct a set of parallel matrices for comparative purposes.

4. The priorities in the various matrices are summarised and compared in terms of weights. The weights can be quantitative and/or subjective based purely on expert judgement for cases where quantification is not possible.

2.6.3 Quantitative methods in business management

Quantitative methods that are used in business involve the use of financial calculations, analysis methods and models. These methods aim to determine the influence that business decisions could have on the financial situation (statements) of an organization. Decisions are strongly influenced by changes that could be made on the balance sheet and the cash flow statement (Libby et al., 2004). Decisions that would influence the financial statements are usually current decisions characterised by short term (quarterly or annual) effects. Decisions that would aim future prospects that are associated with investments are determined by the net present value (NPV) or the internal rate of return (IRR) of financial prospects (Garrison et al., 2006).

2.6.3.1 The net present value method

The NPV method is useful in preference decisions to select the best from competing alternative options. It accounts for *the time value of money* that states that a rand today is worth more than a rand in the future. Projects that provide earlier returns are more favourable than those providing the same returns later. The capital budgeting methods that account for the time value of money are based on *discounted cash flows* (Garrison et al., 2006). It makes provision for the effects of appreciation and inflation based on (Wisniewski, 2002; Silbiger, 1999):

$$NPV = (R\ in\ \ future) \times (1 + discount\ rate)^{-Number\ of\ \ periods} \qquad (2\text{-}1)$$

The discount rate could consist of inflation plus a component that could indicate the risk of the project or investment. Projects will typically be evaluated based on the highest NPV while a negative NPV would indicate unfavourable projects or options.

In environmental project components, there is usually no income to the project, but rather expenses. In cases where expenses are evaluated, the concept changes to Net Present Cost (NPC) in which case the option with the lowest NPC would be more favourable from a financial perspective. Financial decisions may choose between total or cumulative cost and cash flow. In the graph below (Figure 2-6), a water supply option is evaluated for a new development. Option A1 has the lowest total cost, but is not the most favourable in terms of NPC as it has the highest negative impact on cash flows in 2010 and 2011 (Figure 2-6; AGES, 2009). This means that the option with the lowest cost or highest income may be less favourable to an option that has the lowest impact on cash flow or that yields the earliest returns in terms of cash flow (Garrison et al., 2006).

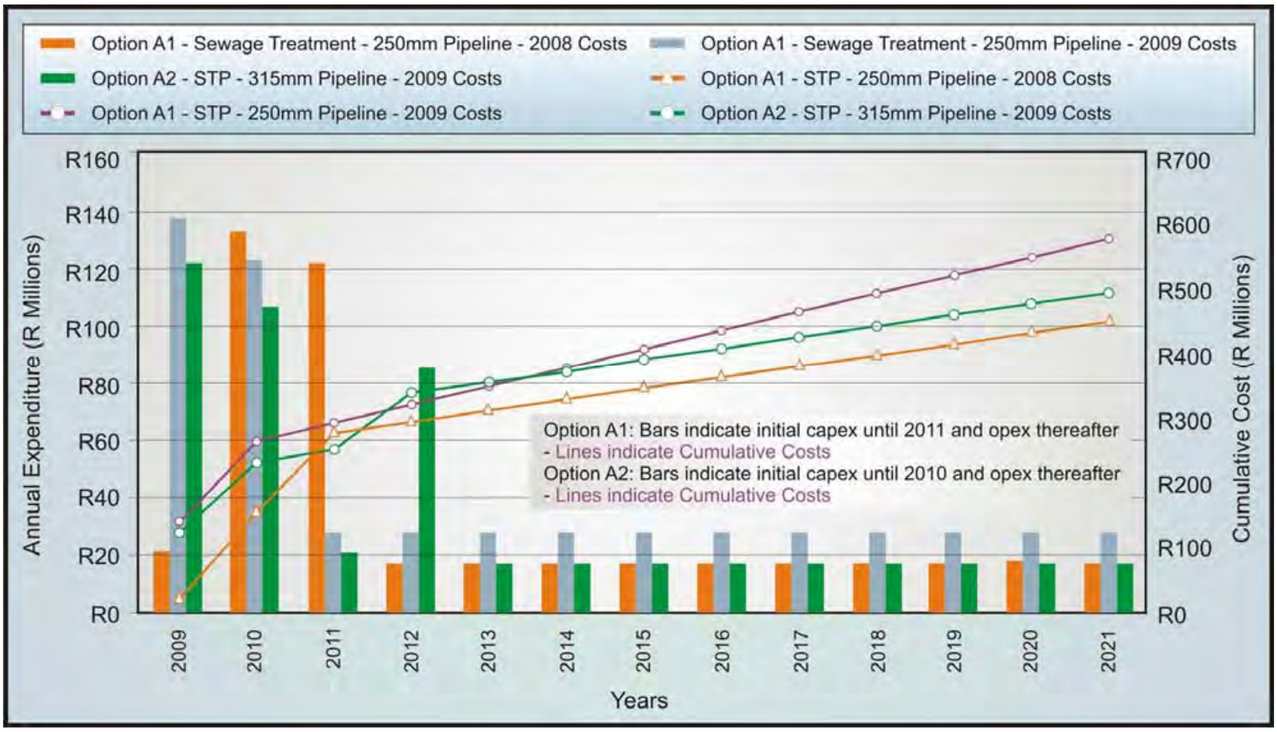

Figure 2-6 Cash flow cost graph showing the cash expenditure with time and the cumulative cost (AGES, 2009).

2.6.3.2 The internal rate of return method

Another important financial decision-making parameter is the internal rate of return (IRR). It is a derivative of the NPV and represents the rate of return promised by an investment project over its useful life represented by (Garrison et al., 2006; Wisniewski, 2002):

$$IRR = \frac{Investment\ required}{Net\ annual\ cash\ \ flow}$$

If investment options are ranked, the one with the highest IRR would be the most favourable as it also accounts for cash flows.

Environmental decisions such as the closure costs of operations such as mining activities and implementation of management and mitigation measures are strongly influenced by the NPC and impacts it could have on financial statements, future capital expenditure and on cash flow. This aspect will be discussed further in Section 2.10.8.

2.6.3.3 Cost benefit analysis method

Cost benefit analysis (CBA) is a business decision tool that determines the range in costs and benefits that influence decision options (Strydom and King, 2009). It is often used in environmental assessments to determine whether a development would have more benefits than costs. CBA is a very useful decision-making tool as it shows and ranks the options, costs and or benefits to the decision-maker.

In Table 2-1, four groundwater remediation options are listed against the cost for the same efficiency, but for different time frames. It indicates to the decision-maker that the Phytoremediation[6] (Option 1) would be the most cost and time effective with a capital cost of R30 million and an operational cost of R3 million per annum. Apart from the cost efficiency, the simulated time frame to capture the seepage of contaminated water would be 20 years against 40 years for the next best Option 4 (Vivier and Gouws, 2002).

[6] Phytoremediation is the remediation of the environment using plants. In this case, the option to use trees to capture seepage of contaminated groundwater was evaluated.

Table 2-1 Cost-benefit analysis of a groundwater remediation options – ranked in preference (Vivier and Gouws, 2002).

No	Description	Limitation on depth to groundwater		Capital costs				Operational costs		Estimated time (uncertain)
		Min	Max	Unit	Cost/unit (R)	Units required	Total (R)	Failure rate (%)	Cost/unit/ year (R)	(years)
1	Phytoremediation	0.5	20	ha	R 15,000	2,000	R 30,000,000	10	R 3,000,000	20
2	Capturing boreholes	5	>100	No of boreholes	R 400,000	100	R 40,000,000	10	R 4,000,000	50
3	Capturing trenches, drains	0.5	4	m	R 3,668	15,000	R 55,000,000	10	R 5,500,000	uncertain
4	Injection-abstraction boreholes	5	>100	no	R 400,000	140	R 55,000,000	10	R 5,500,000	40

2.7 Decision-making in judicial processes

2.7.1 General

Decision-making in the legal sphere is important as environmental regulation is done using legal processes. Although there are some companies and individuals that operate above the legal limits, we have to accept that there will always be those that will try to get away with the least in terms of cost and hence of sustainable development. Environmental legislation has proved to be critical for sustainability. For this reason, it is important to determine how decision-making is done from a legal perspective. Scientific and social information is and will be used in legal actions especially in the form of expert witnesses. It is important to understand the similarities and discrepancies in these approaches.

Environmental and water management in South Africa is regulated by legal processes such as the National Water Act (NWA; Act 36 of 1998) and the National Environmental Management Act (NEMA) (Act 107 of 1998). Any process should take into account that if disputes arise, it will have to be able to go through a judicial decision-making process. Technical processes often encounter problems when challenged in this way.

Pollution[7] is legally defined in the National Environmental Management Act (NEMA) (Act 107 of 1998) as contamination due to human action (Glazewski, 2000). The natural occurrences of fluoride in e.g. the groundwater on the contact zone of the Pilanesberg Intrusion Complex (PIC) does not meet the water quality guidelines (AGES, 2008), but because it is of natural causes, it is not defined as *pollution*. If a mining company would mine in the area and the waste material that contains fluoride leach from the mine residue dumps to levels exceeding regulatory criteria, then it is legally considered as pollution.

The purpose of this analysis is to determine the basic principles of decision-making in judicial processes and how it can be used to enhance sustainability in environmental decision-making.

[7] An undesirable change in the physical, chemical, or biological characteristics of the natural environment, brought about by man's activities (www.answers.com).

2.7.2 Types of legal cases and evidence in judicial proof

In judicial processes there is a distinction between two types of cases, namely criminal and civil. In general, criminal cases require *proof beyond reasonable doubt*[8] and in civil cases, there is a gradation of proof (i.e. probability) and guilt can be split between two or more parties. In criminal cases, a deterministic analogue approach and in civil cases, a probabilistic analogue approach is followed. One can argue that deterministic situations are *idealistic* and actually do not exist in nature. Even the most accurate natural phenomenon will have variations and be probabilistic. In criminal and civil cases, the onus lies with the plaintiff to supply the proof. This is known as the *burden of proof* and the defendant is *presumed innocent until proven guilty*. The plaintiff e.g. has to show a causal link between the defendant's actions and e.g. damages caused. Beyond reasonable doubt does not mean a 100 % proof or certainty. There is no formal definition, but research has shown that jury members asked to give a definition of beyond reasonable doubt *feel* that it is with a probability of greater than 90% (Connoly, 1978) with some that argue a higher figure of 99% (Gardner-Medwin, 2005).

There is an argument that judicial proof is different from mathematical proof, but that mathematical proof could be very useful in judicial proof. This is because there is an inclination to hold that a particular conclusion is doubted because there is a specific reason for doubting it rather than to doubt the level of mathematical certainty (Cohen, 1977). Specific problematic aspects in mathematical probability, is the principle of negation, where the use of assumptions and the superposition or multiplication of probabilities cause legal discrepancies (Time Magazine, 2010, Cohen, 1977). Absolute proof that would be defined in mathematical terms as 100% certainty is rare, but in watertight cases, there could be more than one exhibit that provides 100% certainty, for instance where there are say 3 eye witnesses to a crime. In such a case, it would not make sense to conclude that the defendant is (mathematically) 300% guilty. Cohen (1977) concludes that judicial and mathematical proof differs in definition, but that it could complement each other if used in the correct perspective.

[8] The standard that must be met by the prosecution's evidence in a criminal prosecution, that no other logical explanation can be derived from the facts except that the defendant committed the crime, thereby overcoming the presumption that a person is innocent until proven guilty. If the jurors or judge have no doubt as to the defendant's guilt, or if their only doubts are unreasonable doubts, then the prosecutor has proven the defendant's guilt beyond a reasonable doubt and the defendant should be pronounced guilty. Beyond a reasonable doubt is the highest standard of proof that must be met in any trial. In civil litigation, the standard of proof is either proof by a preponderance of the evidence, weighted probabilities or proof by clear and convincing evidence. (http://legal-

2.7.3 Probability theory and judicial processes

Water science and engineering is based on quantitative methods to delineate, evaluate and understand water resources (Basson et al. 1994). The main purpose is to determine water quantities or qualities to be able to plan ahead or to design a process. There are two types of quantitative, technical processes that can be followed in water sciences. The first is a deterministic approach where exact determinations are made. The output is binary meaning yes or true is represented by a one (1) and no or false by a zero (0). The second is to follow a probabilistic (also known as statistical or stochastic) approach where there are gradations of probability between zero and one, which originated from analyzing odds in games of chance (Olkin et al. 1978; Basson et al. 1994; Mendenhall et al. 2006; Dawid, 2001). A probability of 0.5 that it would rain tomorrow means that there is a 50% chance of rain. Due to the high degree of variability related to natural and technical processes, use is mostly made of probabilistic methods to evaluate data and put answers in perspective in water and environmental resource assessments.

Judgement processes used in the court of law follows a completely different decision process from the technical processes. As mentioned earlier, the problem is generated by the mathematical principle of negation where:

$$Pm(S) = 1 - Pm\ (\text{Not } S) \qquad (2\text{-}2)$$

The mathematical analysis implies that in civil cases, the judicial system is prepared to tolerate a substantial mathematical probability of say 25% that a losing defendant deserved to succeed. An example of this problem is illustrated in *the paradox of the gatecrasher* (Information box 2-A; James, 1941; Cohen, 1977; Rhee, 2007). This indicates that there is a principle based discrepancy between probabilistic assessments, which is the basis of technical assessments and judicial proof (Lindley, 1991).

dictionary.thefreedictionary.com).

What is required as a frame to the pure mathematical process is a solid support and perspective for the probability theory. A process of elimination to verify which facts can be connected with certainty and what must be done to verify the confidence in the probabilities determined or to reduce the uncertainty, is what known as inductive probability (Cohen, 1977). A reference class or policy that can put the probability into perspective would also reduce the problem cited in the contradiction (Rhee, 2007). A policy would e.g. be a guiding principle that in the case of the gate crasher, probability theory could be used to determine that most people are likely to be guilty of gate crashing but that no claim can be installed without actually knowing which ones are guilty or the organizers must be held liable for not issuing tickets.

Another specific gap exists between science and judicial processes where scientific theories, parameters or assumptions are viewed as fact. A good example is when Darwin's theory, after being interrogated by a legal specialist has been shown to have no scientific proof and equated to a religion based on belief. From a legal perspective, there was never any evidence found for support in inter-species[9] evolution, but only for intra-species evolution. The end result is that a belief system developed around Darwin's theory (Johnson, 1991). It is argued that the theory of survival of the fittest cannot be applied everywhere, as it specifically fails the morality test. In a human context, it is not acceptable for one race to kill off another based on the theory of survival of the fittest, as Hitler misused in the Nazi party to support his ideologies (http://www.straight-talk.net/evolution/hit.htm).

Information box 2-A:

The Paradox of the Gatecrasher (Cohen, 1978)

Suppose the threshold of judicial proof in civil cases were interpreted at the level of a mathematical probability of 0.501. Would judges accept the mathematical probability that the unsuccessful defendant that deserved to succeed could be as high as 0.499?

An example to illustrate this point is a fictional situation in which 499 people paid for admission to a rodeo and that 1000 are counted in the seats of whom person A is one. If there were no tickets issued, there would be no way to determine whether A paid for the ticket or climbed over the fence. By any plausible criterion of mathematical probability, there is a mathematical probability of 0.501 that A did not pay for

[9] Inter-species evolution means that a mouse and an ape have the same forebearer etc. Intra-species evolution means that a mouse can change into a bigger or smaller one etc, but are not necessarily related to other species.

admission ($Pm = \frac{(N_{people\ in\ stand} - N_{tickets\ sold})}{N_{people\ in\ stand}}$). Based on this probability, the organizers would be entitled to claim the money from A, since the balance of the probability is in their favour. It seems unjust that A should lose the case if there is a mathematical probability as high as 0.499 that he did pay for admission. If the organizers would be really entitled to claim against A, they would be equally entitled to make a claim against each person at the rodeo, thereby recovering 1000 admission moneys, when it was admitted that 499 did in fact pay. This leads to a scientific contradiction and an absurd justice that shows something must be wrong.

It must be accepted that in practice, *de facto* injustice will be done in even the best of judicial systems. The probability under discussion does not take these circumstances into account. It demonstrates the limitations of a pure mathematical probabilistic approach.

For other illustrations of the same problem, see the paradox of the blue bus (Redmayne, 2006) and Trial by Mathematics (Time Magazine www.time.com/time/magazine/article/0,9171,838296,00).

The gap between the legal and scientific processes in environmental management could cause problems for sustainable development or management (Figure 2-4). The technical or legal processes could contradict each other which would mean a much more difficult road to sustainability. A case in point is that no South African company has ever been found guilty of *pollution* in a court of law (Cross, 2009), while there are known instances of pollution. Legal aspects would form constraints within which environmental decision-making should take place. It also acts as an acid test for scientific arguments.

Legal decision-making processes have known limitations. Research has shown that the way in which a question is framed can lead witnesses to answer differently to the same facts (Loftus, 2000), the way or sequence in which evidence is portrayed determines jury decisions (Hastie and Pennington, 2000) and that political interferences do occur.

2.7.4 Judicial decision-making principles

Decision-making in judicial processes are based on principles and values rather than scientific probabilities. Although scientific probabilities might be used in legal cases, the merit based on values would be more influential (Viljoen, 2010). Some of these principles in South African and international (environmental) law are (Cohen, 1977; NEMA, Act 107 of 1998; Glazewski, 2000):

1. Innocence until proven guilty. Any accused in a criminal case will be presumed innocent until proven guilty.
2. Proof beyond reasonable doubt for criminal cases and balance of probabilities for civil cases. Causal effects between the accused and actual damages must be proven.
3. Burden of proof is on the accuser. The onus is on the accuser to prove that the accused is guilty. Internationally, there have been calls for a reversal in the burden of proof principle in environmental law. The basis of this principle is that the party attempting to preserve a less polluted environmental state should not carry the burden of proof (Sahasranaman, 2009).
4. Data, information and conclusions are cross-examined and interrogated by parties taking both sides of the argument to determine what qualifies as evidence[10].
5. A case is built for or against an argument. The case that weighs the most usually wins.
6. Polluter pays principle. If found guilty, the polluter must be held liable for financial loss or costs incurred. These could involve future costs or lost opportunity for income (Section 2.6.3).
7. Precautionary principle. The inadequacies (i.e. uncertainties) of science have led to the adoption of the precautionary principle. There are five components that should guide the implementation of this principle (Sahasranaman, 2009):
    a. Action to prevent harm despite uncertainty.
    b. Shifting the burden of proof to potentially harmful activity.
    c. Examination of a full range of alternatives to potentially harmful activities, including no action.
    d. Democratic decision-making to ensure inclusion of affected parties.

[10] The facts, signs or objects that make you believe that something is true (Oxford English Dictionary, 2006). Data on which to base proof or to establish truth or falsehood (Collins English Dictionary, 2006). Direct evidence is a real, tangible or clear evidence of a fact that requires no consideration to prove its truth or existence. Circumstantial evidence is information and testimony presented by a party in a civil or criminal action that permit conclusions that indirectly establish the existence or nonexistence of a fact or event that the party seeks to prove (http://legal-dictionary.thefreedictionary.com).

e. Where decisions that could involve potential environmental impacts should be made in favour of or conservatively to protect the environment.

The principle of the burden of proof that is on the accuser could be considered as a problem. Environmental investigations are very expensive and the impact of environmental pollution is mostly on surrounding communities that do not have the financial means to conduct investigations to the level of proof beyond reasonable doubt. This principle is called for review by regulatory authorities.

2.8 Decision-making in the political sphere (social)

2.8.1 General

Environmental and sustainable development today receives focus in the global arena and are in cases highly politicised (Gore, 2006). The media exposure that green aspects receive is at a high point and is usually linked to world economic forums. Political motives are well developed and legislated in South Africa. It can strongly influence or even override all the other disciplines or facts with positive or disastrous consequences. The influence that politics have on environmental decision-making is critical and should therefore be accounted for in environmental decision-making.

2.8.2 The political decision-making model

The political model for decision-making represents the self-interests and goals of powerful external and internal stake holders. Power is the ability to influence or control individual, team, departmental and organizational decision-making and motives. Political processes are most likely to be utilised when powerful stakeholders and decision-makers disagree over the most important goals, especially when analysts (technical, business or environmental) do not succeed in reaching acceptable or suitable alternatives (Hellrieger, et al. 2002).

The political model accounts for a divergence in goals that is likely to lead to conflict between stakeholders. When conflict arises, it also leads to a divergence in solutions. There is often no clear winner and if there is one, it leads to a win-lose situation. Political decision-making models could be advantageous when stakeholders are willing to negotiate a win-win situation. This happens when the parties concede that everyone will not be able to aim for the best option of individual stakeholders, but for the best option for the cause. A balance of power usually leads to a negotiated solution and is characterized by situations where alliances are formed (Hellrieger, et al. 2002).

In the political model, people are individually but not collectively rational (Section 2.6.1). The political model becomes more important as the size of organizations gets bigger. Organizations are coalitions of people with competing self-interests. Power struggles become evident when use is made of tactics such as coalition formation, co-optation, strategic and selective use of information and the use of outside experts (Eisenhardt and Zbaracki, 1992).

The environmental field in South Africa is highly influenced by political powers. All environmental regulatory authorities are large politicised organizations. The NWA has a large political component for redress of previously disadvantaged groups, to the extent that concerns are raised of double standards such as less stringent or no control effected on municipalities (local government) on e.g. waste water discharge and landfill site management. Environmental impacts of low income developments or developments where job creation is important for the local (political) community are sometimes seen as less important than e.g. mining and other developments initiated by rich income groups.

Politically charged organizations do not operate functionally. Appointments are made based on political rather than technical or competency-based qualifications or experience. This leads to *organized anarchies* or *garbage can* organizations (Eisenhardt and Zbaracki, 1992). A problem in South Africa is that a number of regulatory departments could be classified in this category, which leads to inefficiency and biases on environmental regulation. This leads to environmental processes that are not evaluated or decided upon and it could take years for regulatory documents (such as water use licenses) to be processed. It then leaves the developers with no choice but to operate in the absence of approved licenses.

Political influences are very important as it can either strengthen or hamper sustainable development. In most cases, it aims to please the majority of the people and not the basis of the cause.

2.9 Decision-making in science and engineering (quantitative)

"There are no facts, only interpretations."
- Friedrich Nietzsche (1844-1900)

2.9.1 General

The word *science* originated from the Latin word *scientia* that means *knowledge*[11] (www.sciencemadesimple.com). Decision-making in science and engineering is based on quantitative methods, measurable or experimentally *proven* theories with acceptable assumptions. Since the inception of the information age, the computational capability of computers has made the use of quantitative methods and models accessible to everyday use. These models assist in the determination of numerical values or statistical distributions as outcomes that could be valuable in the decision-making process. Problems are often encountered when the correct scientific methods and models are used in the wrong context or without a decision-making method or process that provides the framework (Kozak, 1994; Van Blerk, 2000).

In this section, the use of quantitative methods in science and engineering with reference to decision-making is discussed.

2.9.2 The philosophy of science and the scientific method

To understand the methods that are used in science, it is necessary to consider the origin of science. It is also important to know what the boundaries of science are and what is intended with a scientific process.

Science originated from religion and philosophy. Philosophical science was introduced by Socrates around 400 B.C. His legacy lived on in Plato and Aristotle until the Scientific or Copernican Revolution where it was through observation where it was determined that the Earth is not at the centre of the universe (Gorham, 2009). The big breakthrough and change for science was in the 1600's when Isaac Newton formulated his Mathematical Principles of Natural Philosophy (known as Principia) in 1687. In his work, Newton described his three laws of motion and universal gravitation (Gorham, 2009).

[11] Knowledge is the information, understanding and skills that are attained through education or experience (Oxford English Dictionary, 2006). The facts or experiences known by a person or a group of people (Collins English Dictionary, 2006). Note the

Initially, science was defined by testable or experimentally provable natural phenomena that can usually be described by mathematics. Science evolved and today it consists of *hypotheses*[12], *theories*[13] and *experimental proof* (Figure 2-7). When a theory has been tested and accepted it can also be termed a *scientific law*. Scientific discoveries often start with an idea, which is formulated into a hypothesis that is accepted as a theory when it is experimentally proved or universally accepted (Gorham, 2009). The scientific method assists to organize thoughts and procedures in an organized way (Figure 2-7). Scientists make use of thoughts and/or observations that lead the formulation of hypotheses and deductions to reach conclusions (`Tyler Miller, 2005; www.sciencemadesimple.com). An important component for science is to make the link between *cause and effect* through experimentation or calculation (www.sciencebuddies.org). In this aspect, science and judicial processes are similar (Section 2.7.4).

> "It has often been said that the greatest discovery in science was the discovery of the scientific method of discovery." --Dr. James K. Feibleman, author of Scientific Method (1972).

Science is based on reasoning and the quality of the reasoning depends on the connection between the information and the conclusion. Logically, there is a fundamental distinction between two forms of scientific reasoning known as deductive and inductive reasoning (Gorham, 2009). If the information that is relied on in good deductive reasoning is true, then the conclusion is guaranteed to be true. It corresponds to *direct evidence* in law (footnote 10). An example of this is; that if a statement is made that all sheep are herbivores and all herbivores are animals, then all sheep are animals. This can also be termed the deterministic or binary (0 or 1) approach where a statement is either true or false (Section 2.7.2).

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difference between knowledge and wisdom (footnote 2). Wisdom is of a higher level and encapsulates knowledge.

[12] An idea of explanation of something that is based on a few known facts, but that has not yet been proven to be true or correct (Oxford Dictionary, 2006).

[13] A formal set of ideas that is intended to explain why something happens or exists (Oxford Dictionary, 2006).

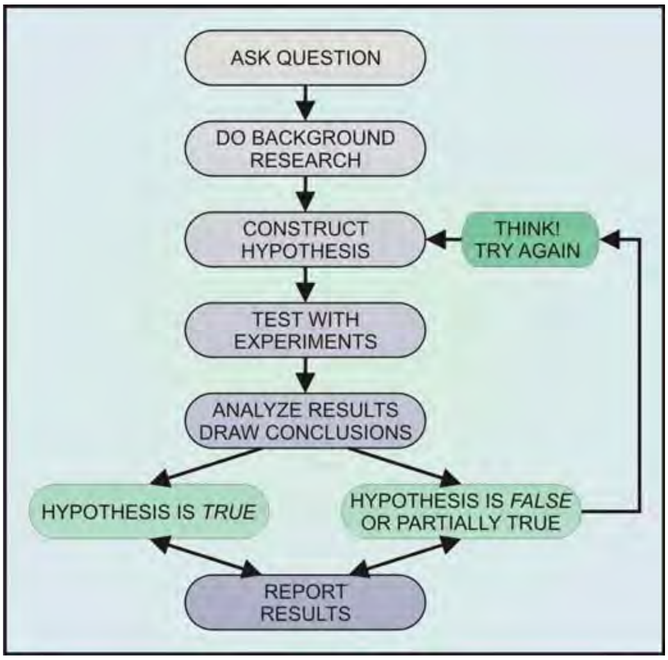


Figure 2-7 Schematic representation of the Scientific Method (www.sciencebuddies.org).

In the case of good inductive[14] reasoning, the truth of the conclusion is not guaranteed, but based on its premises the truth is likely. It corresponds to *circumstantial evidence* in law (footnote 10). An example of this is; thousands of sheep species have been observed that have tails, so all species of sheep are expected to have tails. If the first statement is true, then the second is probable. This can also be termed the probability (i.e. statistical) approach in which gradations of probability exist and not an absolute true or false (Section 2.7.2). In inductive reasoning, the conclusion reaches beyond the known facts, which often led to new scientific discoveries (Gorham, 2009). Scientists use inductive reasoning to convert observations and measurements to generalized conclusions and deductive reasoning to convert generalization to a specific conclusion (Tyler Miller, 2005).

[14] Inductive reasoning has often led to new discoveries. In 1846, Johann Gottfried received a letter from a Frenchman, Urbain Le verrier who had been studying the motion of Uranus that its path could not be explained by the known gravitational forces acting on it. Galle went to his telescope and discovered the planet Neptune (Feng and Trodden, 2010).

An exhaustive debate still exists in scientific philosophy on when a theory can be classified as scientific. This problem was addressed by Karl Popper as *a theory is scientific to the extent that it is falsifiable by means of experimental tests*, which he used to distinguish between *science* and *non-science* (Stanford Encyclopaedia of Philosophy, 2010; Hutcheon, 1995). According to him, the requirement of pure observation as an initial step for the formulation of a scientific theory is misguided as all observation is selective and influenced by the theory. He denotes that it is easy to obtain evidence in support of any theory (Stanford Encyclopaedia of Philosophy, 2010). Popper considers a theory as scientific only if it is refutable by any conceivable event and that verifications of the theory would strengthen its case. Logically, a scientific theory is conclusively falsifiable but not conclusively verifiable. No observation or experiment is error free, so any experiment or observation can be questioned. Einstein's theory of relativity was considered by Popper as scientific as it was highly falsifiable. Similary, Darwin's theory of evolution was not considered as scientific as it is not falsifiable.

The origin of science stems from problem-solving and not only from observations. Science is therefore defined more by its method than by its subject matter (Gorham, 2009). The notion that there is something like an *exact science* is therefore not true. Science is *not exact*, but stands on the most probable outcome based on experience or probable theories. Popper refers the term *normal science* and does not make reference to *exact science* as an exact science does not exist (Stanford Encyclopaedia of Philosophy, 2010).

Newton's laws were considered as the truth where mass and time were absolute until Einstein's theories indicated it to be relative (Einstein, 1916; Kaku, 2004). Einstein's theory could only be proven much later in the 1960's when hydrogen clocks could be measured against each other with one stationary and one in a supersonic jet that was flown around the world. This proves the power of allowing theories and not only facts as scientific basis. Science is therefore based on hypotheses and theories that can/will be proven wrong or *not exactly accurate* on an ongoing basis (Gorham, 2009).

Analogies to conditions or substances that do not (and cannot) exist or cannot be proven yet, are often used in science to demonstrate certain principles. The use of the *ideal or perfect gas law* in physics is an example of this (Beuche, 1986). The use of idealizations in scientific reasoning to distinguish between concepts has proved to be useful in decision-making. Scientists therefore do two things, they can disprove things and it can be determined that a particular aspect or model has a high probability of being true. Scientifically, no theory can be proven as absolutely true (Tyler Miller, 2005).

2.9.3 Data and information

The role of data and information in the decision-making process will be discussed in the next chapter (Section 3.2).

2.9.4 Statistical methods in scientific decision-making: The Bayesian approach

As discussed in the previous section, there are two basic types of quantitative assessments. The first is deterministic or deductive that considers facts to be either true or false and the second statistical or inductive that considers gradations of probabilities (Sections 2.7.3 & 2.9.2). In environmental and water management, statistical methods are considered a more appropriate approach, as most natural environmental processes such as rainfall, evaporation etc are described by variability and probability (Basson, et al 1994). The statistical patterns or distributions can be analysed which are then used in resource quantification and prospective evaluations (*also incorrectly known as forecasting or predictions*).

There are two schools of thought in statistics namely, classical statisticians or *frequentists* and the *Bayesian* statisticians (Du Plessis, 2009; Freeze et al. 1990; Freeze et al. 1992; Ellison, 1996). Classical statistics require a basic minimum set of data that is used to determine statistical variables such as the mean, the median, standard deviation etc (Wisniewski, 2002). There seems to be a large rift between Frequentists and Bayesian statisticians as the frequentist approach requires a base minimum of sample data before evaluations can be made (Du Plessis, 2009). The conundrum using the frequentist approach, is that is does not provide answers in cases where data is sparse and uncertainty high, which is mostly the case in groundwater and environmental water management.

Bayesian statistics accepts that statistical variations such as the mean, median, standard etc. can be inferred based on known information or a prior judgement (Figure 2-8). In probability theory, the Bayes Theorem indicates how one conditional probability is dependent on its inverse. The probability of event A (groundwater head elevation), given or conditional to event S (topographic head elevation) does not only depend on the relationship of A and S (i.e. the accuracy of the topographic elevation), but of the absolute probability of P independent of A (Standford Encyclopaedia of Philosophy http://plato.stanford.edu/entries/bayes-theorem; Zeng and Sycara, 1996). The probability H conditional on E is defined by the Bayes' Theorem as (Mendenhall et al. 2006):

$$P(S_i, A) = \frac{P(S_i)\,P(A, S_i)}{\sum_{j=1}^{k} P(S_j)\,P(A, S_j)} \qquad (2\text{-}3)$$

With *i=1,2,3....k,* given that both terms of this ratio exist and that P (A) > 0. Analogue data can initially be used to estimate the unknown parameter in what is known as a prior analysis that is updated iteratively with an increased level of certainty using a posterior analysis (Figure 2-8; Freeze et. al., 1990; Kitanidis, 1986; Van Sandwyk et. al. 1992; Ming et al. 2004). The approach followed in this study would be to make use of the frequentist approach if data points are sufficient but to use the Bayesian approach where data is sparse (Ellison, 1996; Ming et al. 2004).

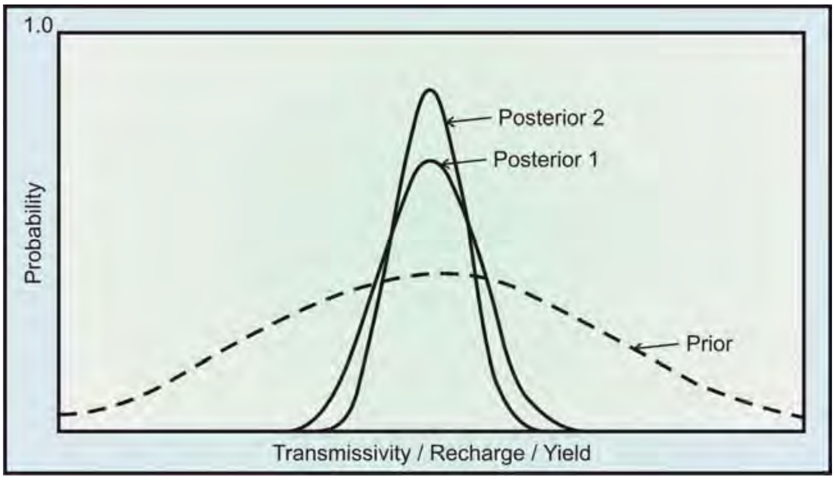


Figure 2-8 Graphic representation of the Bayesian probability prior and posterior analyses (Mendenhall et al. 2006).

A well-known and applied use of Bayesian statistics in geohydrology is the use of known topographical elevations to predict groundwater head elevations (Van Sandwyk et. al. 1992; Buys, et. al. 1995). This is particularly useful in groundwater models to pre-evaluate the spatial distribution of groundwater head data (or contours) because the topographic elevation data of the whole country is available at e.g. 1:50 000 scale. Where a good correlation between topographic elevation and groundwater head elevation exists, the head elevations can be pre-determined by using the topographic elevation data points. The measured groundwater head data from as few as 5 to 15 borehole data points can in this way be interpolated across the model domain using the Bayes approach (Figure 2-9).

The Bayesian approach is ideally suited for environmental decision-making, such as water management. The Bayesian approach can avoid the proverbial *doing research while the building is on fire* problem that occurs when sufficient data is collected, it may be too late to rectify the problem.

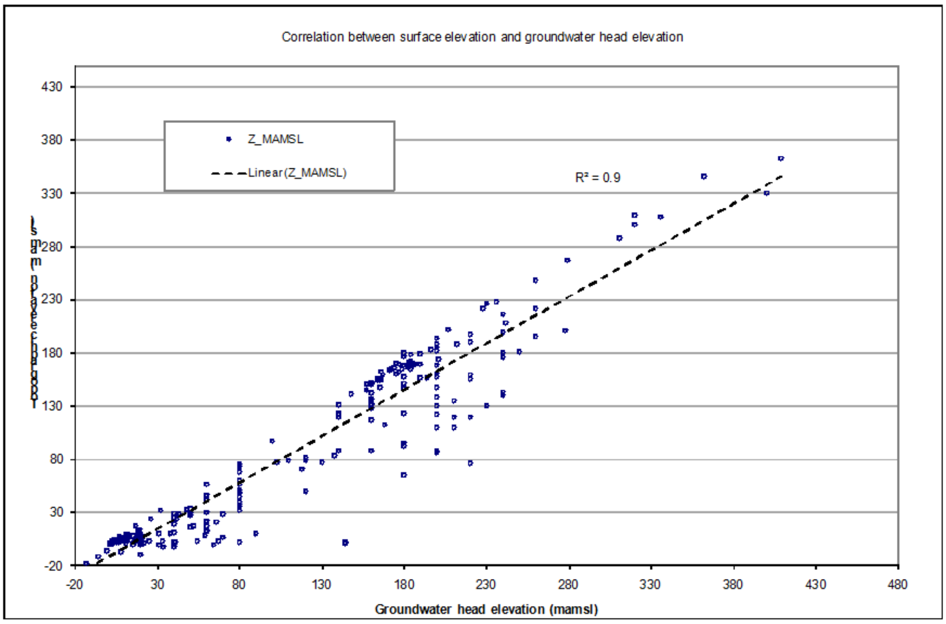


Figure 2-9 Graph showing the correlation between topographic and groundwater head elevation (Vivier et al. 2009).

2.9.5 The decision tree

A decision tree consists of a graphical representation of the problem with options and probabilities that are weighted (i.e. balance of probabilities). It can be used to map out probability problems and assign probability values to a series of options. The advantage of a decision tree is that it can be done at various depths or levels so that downstream consequences and probabilities can be evaluated (Wisniewski, 2002). It provides both a graphical representation of the options in the decision-making process as well as the numerical and probabilistic variables. Decision trees can also be used to represent non numerical data such as political aspects as constraints in terms of fuzzy representations (Janikow, 1996). A classic decision tree is analogue to the rationality model for decision-making (Section 2.6.1). A problem that can arise with decision-trees is that it could become overly complex, especially when higher order levels are introduced. Higher order decision trees could also suffocate from computational requirements. Methods, such as pruning and minimax were introduced to cut back on the tree branches and focus on the most important branch options (Quinlan, 1986). The framed or pruned decision tree corresponds with the bounded rationality model (Section 2.6.2).

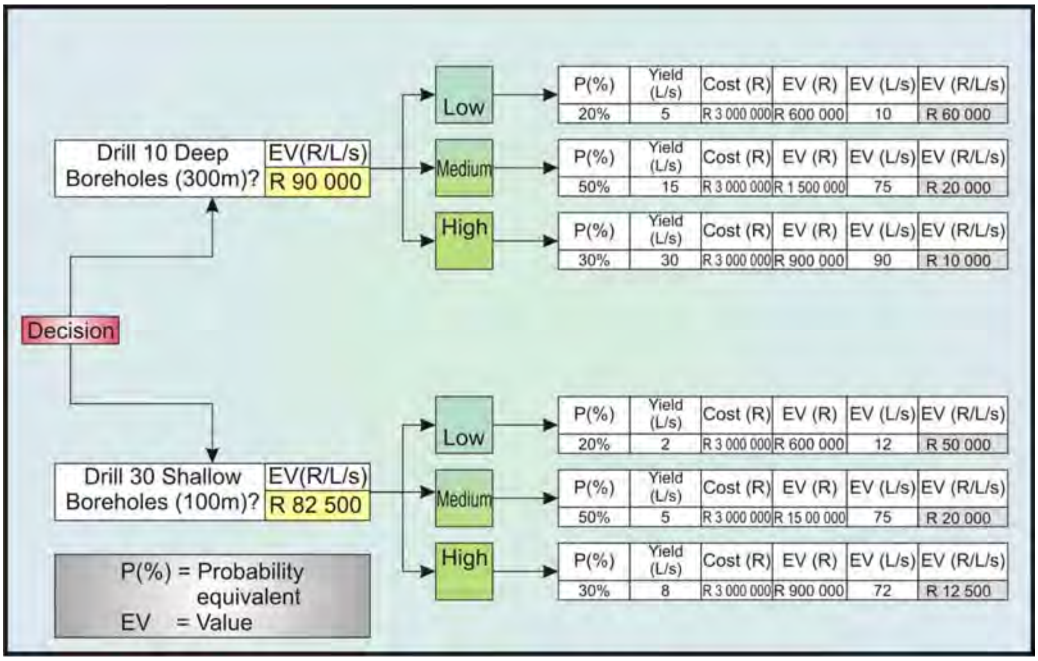


Figure 2-10 Graphical representation of a decision-tree.

To show the usefulness of a decision tree, an actual example from a groundwater problem was used. A decision had to be made to drill either 10 deep boreholes (300 m deep) costing R300 000 per borehole or 30 shallower boreholes (100 m deep), costing R100 000 per borehole. The budget constraint for drilling was R 3 million. Additional information was that the deep boreholes are expected to yield between 5 ℓ/s and 30 ℓ/s per borehole, while the shallower boreholes are expected to yield between 2 ℓ/s and 8 ℓ/s per borehole. The cost and yield of the boreholes were determined in the decision tree using the equivalent value (EV). The EV was determined as the probability times cost and the EV for cost and yield were determined as R/ℓ/s to provide a unit EV. The decision tree shows that although the difference is effectively small, the lowest EV of R82 500/ℓ/s was expected for the shallower boreholes against R90 000/ℓ/s for the deep boreholes. The shallower boreholes were chosen from a cost consideration.

2.9.6 Artificial intelligence, game theory and the chess analogue in decision-making

Game theory is the theory of independent and interdependent decision-making and represents a theoretical model of reality. The role players could be individuals in an organization or the natural environment. It represents an abstract model of decision-making and not the social reality of decision-making (Kelly, 2003). Games like chess and bridge fall within the ambit of game theory but so does organizational and technical decision-making. There are a number of game categories. Some are cooperative that leads to win-win situations where negotiated settlements are reached (Section 2.8.2), some are competitive that leads to win-lose situations (also known as zero sum or non-cooperative) like chess or some business cases while some are partially cooperative. All theories and models are simplifications of reality, but should not be dismissed simply because it fails to represent reality. Game theoretic models should be dismissed if its predictions are false or useless (Kelly, 2003). There are three gaming categories namely, games of skill, games of chance and games of strategy. Game theory has developed to a stage where negotiation options can be simulated using a Bayesian learning process (Zeng and Sycara, 1996; Section 2.8.2 and Section 2.9.4).

In this section chess is considered, which is one of the best examples of a game theoretical model for strategic decision-making. It involves strategic planning with tactical changes within a space and time constraint. It has all the characteristics of e.g. an environmental decision-making problem as there is a large number of possible choices (or moves) that can be made based on trade-offs. Like most environmental management problems, there can be no move made without paying a price. It is based on alternative causes of action that has impacts. The objective is to make the move that will cost the least or gain the most.

There are between 10^{40} to 10^{50} possible legal moves in chess, with a game tree complexity of 10^{120}. This is known as the Shannon Number (http://en.wikipedia.org/wiki/Chess). There are 400 different positions after each player makes one move, 72 084 after two moves are made by each player and +288 billion possible moves after each player has made four moves (http://www.chess-poster.com/english/notes_and_facts/.htm). This is known as the combinatorial explosion of decision forks or trees. Chess is for all practical purposes infinite. How is it then possible that chess engines are so good at playing the game (i.e. decision-making)? One can only try to play against a chess engine to admire the decision-making capability. The first time a chess engine could beat a chess world champion was in 1997 when the IBM chess engine known as Deep Blue defeated Gary Kasparov, then the world chess champion. Since then chess engines have improved dramatically as computing power became exponentially faster (http://en.wikipedia.org/wiki/Human-computer_chess_matches). Chess engines conform to the bounded rationality model as it does not aim to evaluate all possible options due to time and practical constraints (Section 2.6.2).

It was found that a computational model of human chess intuition and intelligence can be used to explain decision-making, especially by avoiding the combinatorial explosion (Luger, 2002; Linhares, 2009). Chess engines are some of the best examples of artificial intelligence and are analogue to decision trees that are constructed to determine follow-on moves and combinations that are possible. It is useful as a decision-making training tool that assists the analyst to *see beyond the horizon*. Normal thinking patterns and intuition can cope with linear relationships. It is when the potential outcomes are beyond the horizon, where it is characterized by non-linear relationships that are counter intuitive, when normal thinking patterns and intuition fails. It is here that qualitative and quantitative models can assist in the decision-making process. Chess has been widely used as an educational tool as it enhances the way of thinking as cognitive decision-making patterns develops (Przewoznik and Sozzynski, 2001; Kotov, 2001). Research has shown that students who received chess instruction scored higher on all levels of academic achievement

(Smith and Cage, 2000). It is therefore the philosophy behind chess and specifically chess engines that we want to analyze for the purposes of understanding the decision-making processes. A chess algorithm is based on the following:

1. A definition of the problem boundaries i.e. the 64 squares of the chessboard.
2. A definition of the role players, the boundaries of the role players in terms of movement or influence and weighting of the role players. This would be to define 16 role players on each side, the boundaries of influence would e.g. limit a pawn to a step by step forward movement and provide free reign to a queen. The weighting would be e.g. rate a pawn as having a value of one and a king as infinite.
3. Positional play weighting is important where additional weights would be given e.g. a pawn that has a king in check or which is about to become a passed pawn, would have a much higher weight than any other pawn. Positional play is the most important aspect of the game (Kotov, 2001).
4. Algorithms can be developed to portray a certain problem as a scenario to solve, known as problem or management scenarios (Littman, 1996). Data bases of past experiences can be accessed that determines previous outcomes to reinforce the decision-making process. This was done in the case of the chess engine Deep Blue that defeated Gary Kasparov in 1997. A data base of Kasparov's past games was accessible for the algorithm to predict counter moves (Campbell, et al. 2002). Using past data bases to qualify or calibrate and validate decision-making models is an important component to build confidence in the outcomes for future use.

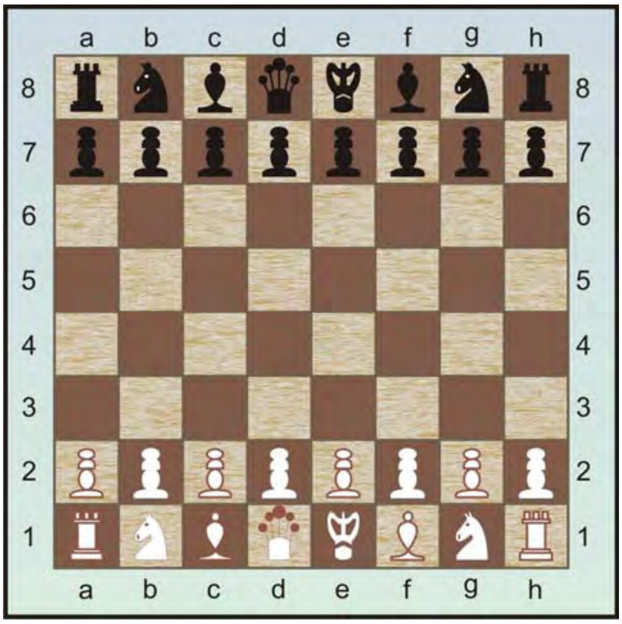


Figure 2-11 A chess board decision domain with initial condition for role players.

2.9.7 Optimization

Optimization involves mathematical programming where the best element is chosen from a set of alternatives and constraints (Walsh, 1979; Kincaid and Cheney, 1991). It is one of the most powerful decision-making tools, especially where multivariate data and/or non-linear trends are involved.

Linear programming is commonly used in optimization of groundwater management problems (Bear, 1979). Advanced optimization is used in environmental cleanup investigations where costs could be very high and the problem and constraints defined in a three-dimensional heterogeneous hydrogeological media leading to non-linear systems (Karatzas and Pinder, 1994; McKinney et al. 1994). The latest development is the use of genetic and artificial intelligence algorithms to solve complex optimization functions in environmental problems that are essentially non-linear (Katsifarakis, et al. 2009).

2.9.8 Scientific models

A *scientific model*[15] can be defined as a conceptual, dimensional or mathematical simplified representation that acts as an analogue of the actual or real world system (Bear, 1979; Van Blerk, 2000; Tyler Miller, 2005). A country map with roads represents a two-dimensional model of an actual country and the roads on it. A roadmap is very useful to navigate, but e.g. cannot be used for outside its design purpose (http://www.stat.columbia.edu/ ~cook/movabletype/ archives/ 2008/06/ all_models_are.html). Another example of a model is an architect plan for a building. Models are useful tools in science and decision-making if applied within the correct context. One of the first models that were used in science was the Bohr model for the atom (Beuche, 1986).

Scientific models usually represent relationships between variables (Hopkins, 2000). The purpose of a model is to pre-determine an outcome of a natural (or economical) process or event before it actually happens, by using prospective evaluations. Models are in essence decision-making tools that should assist the analyst in understanding the problem. Models are the principle tools of modern science. Examples of models are; probing models, phenomenological models, computational models, developmental models, explanatory models, impoverished models, testing models, idealized models, theoretical models, scale models, heuristic models, caricature models, didactic models, fantasy models, toy models, imaginary models, mathematical models, substitute models, iconic models, formal models, analogue models and instrumental models (http://www.StanfordEncyclopaediaofPhilosophy, 2010).

A specific problem with models is when incorrect or unfeasible expectations are raised, based on the outcomes it aims to pre-determine. Models generate some of the biggest controversy in science and engineering. This led to the well-known quote by George Box that "All *models are wrong, but some are useful*" (http://www.skymark.com/resources/leaders/box.asp; Sterman, 2002; Poeter, 2006). When it comes to models, some scientists are what philosophers term *naive realists*[16] who believe from what they see or experience that some scientific things are just plain true or false and that they know what those are (Ross and Ward, 1977; Lehar, 2000). From a

[15] The word model relates to a copy of something, but usually smaller than the original subject (Oxford English Dictionary, 2006). Representation of a device, structure etc, on a smaller scale (Collins English Dictionary, 2006).

[16] Naive Realism is the theory (or expectation) that the world is perceived exactly as it is. Also called natural realism or commonsense realism (http://www.thefreedictionary.com/naive+realism).

modelling perspective, it must be accepted that human knowledge and perception is limited and will change as new discoveries are made (Section 2.9.2; Sterman, 2002).

From the perspective of a naive realist, all maps are *wrong* and should therefore be discarded. The naive realist is caught up in an incorrect pursuit of a determinist accuracy that entails *all* information, which is a situation that does not exist. This aspect is discussed in more detail in Section 5.5.2.

Scientific models usually develop from concept level to analogue and mathematical level, which can be used for prospective evaluations in decision-making. In this section the scientific models and thought processes that are used in resource quantification for the purposes of decision-making are discussed.

2.9.8.1 Conceptual models

Before the development of a model for a specific problem is done, it must be accepted that a problem with no boundaries cannot be solved (Van Blerk, 2000; Figure 2-12). The development of a conceptual (or mental) model is one of the first steps in using models for problem solving or problem understanding. A conceptual model is a description in words and/or diagrammatical / schematical representation of the problem as seen by the analyst. It represents how we perceive and process information[17]. Einstein's thought experiment where he saw himself racing a light beam was a conceptual model that led to the development of the theory of relativity (Kaku, 2004).

Conceptual models are useful in the sense that they are not limited to mathematical limitations. The analyst is free to portray very complex or seemingly impossible aspects in a conceptual model. Conceptual models are often used in geosciences to explain geological depositional environments (Zumberge and Nelson, 1976). These models can either be just graphical representations of a physical system prior to or after a certain development has taken place. An example is a graphical conceptual model to indicate the potential influence that mining could have on a groundwater system (Figure 2-13, Figure 2-14). It can also have a more schematical

[17] http://www.stat.columbia.edu/~cook/movabletype/archives/2008/06/all_models_are.html

representation portraying a system, where initial values are ascribed to determine interactions between different physical components, such as an aquifer and a mining operation (Figure 2-15).

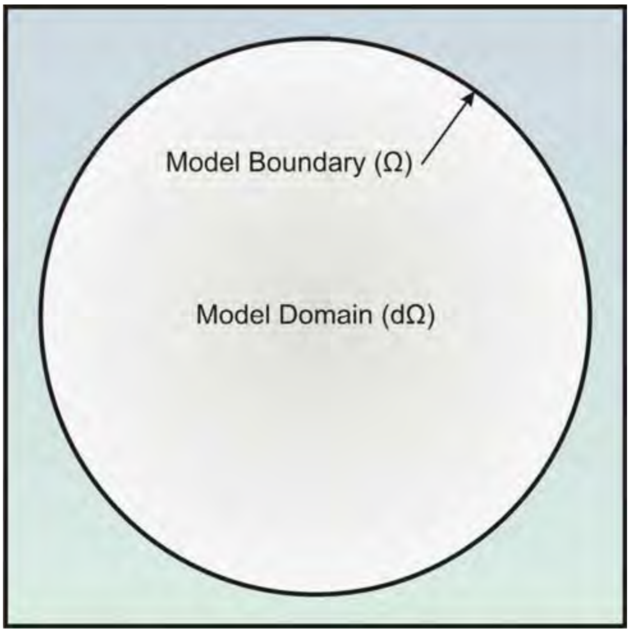


Figure 2-12 Schematic representation of a model (Ω) domain and its boundary (dΩ) (After Botha, 1996).

2.9.8.2 Analogue models

Analogue models are physical models that are built to test the behaviour of a physical system. A model aeroplane is an analogue of an actual aeroplane. A sand box model can be used as a scale model to simulate groundwater flow conditions (Bear, 1979).

2.9.8.3 Mathematical models

Since the advent of especially personal computers, the use of mathematical models proliferated. The development and use of a mathematical model follows the development of a conceptual model. It is usually a reduction of the conceptual or analogue models where the system is simplified using assumptions.

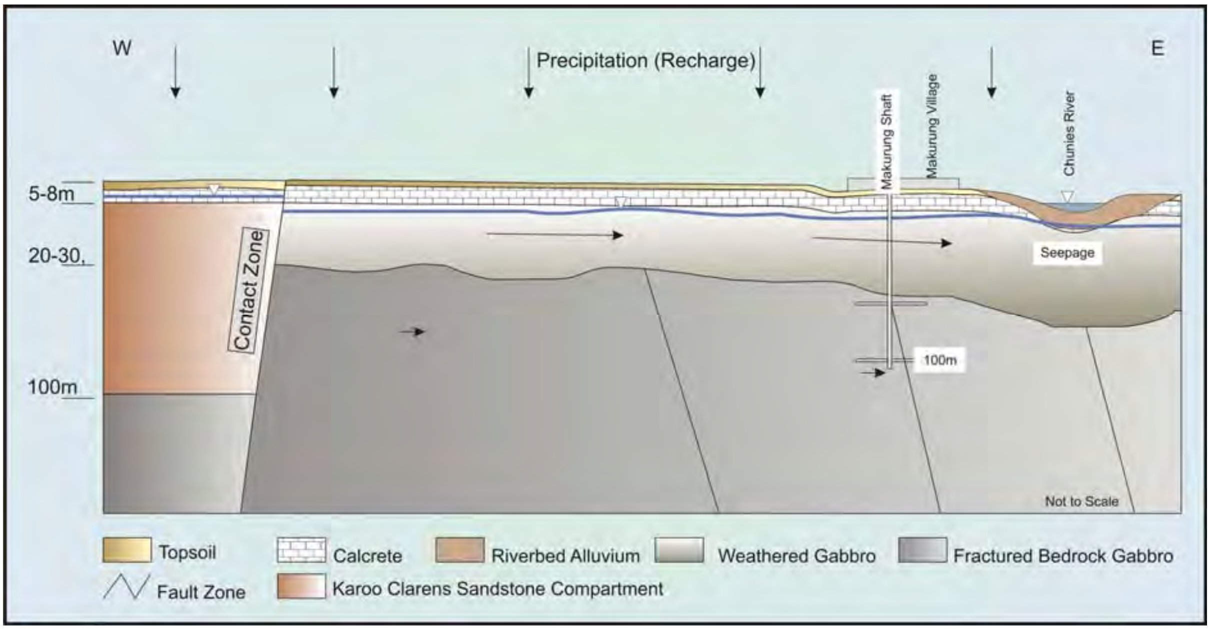


Figure 2-13 Graphic representation of a pre-mining groundwater conceptual model (after Vivier, et al. 2009).

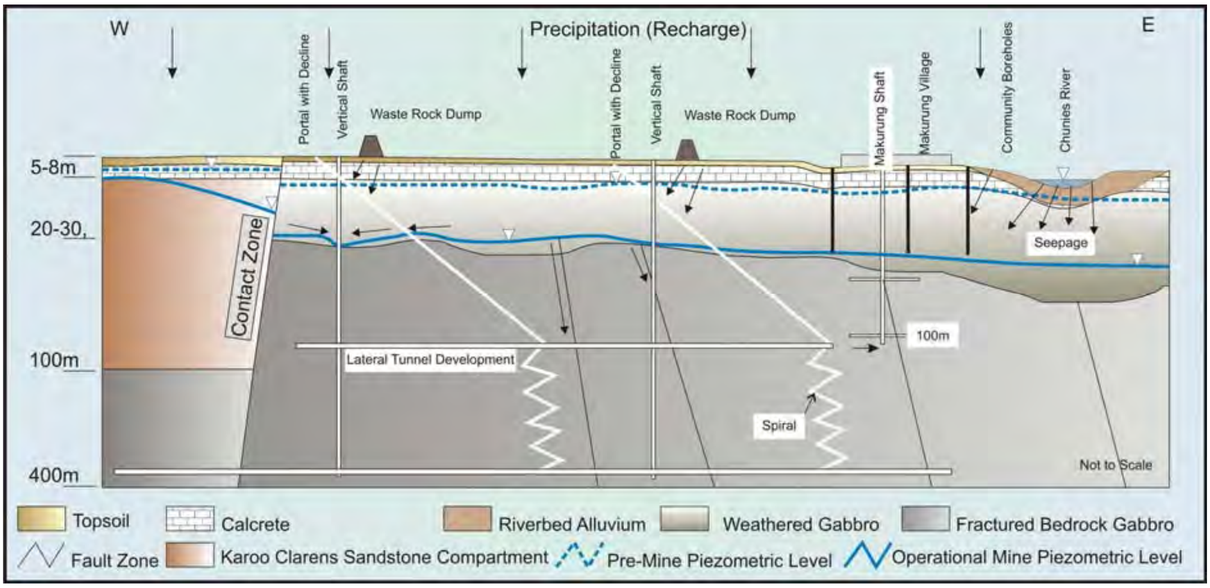


Figure 2-14 Graphic representation of an active mining groundwater conceptual model (after Vivier, et al. 2009).

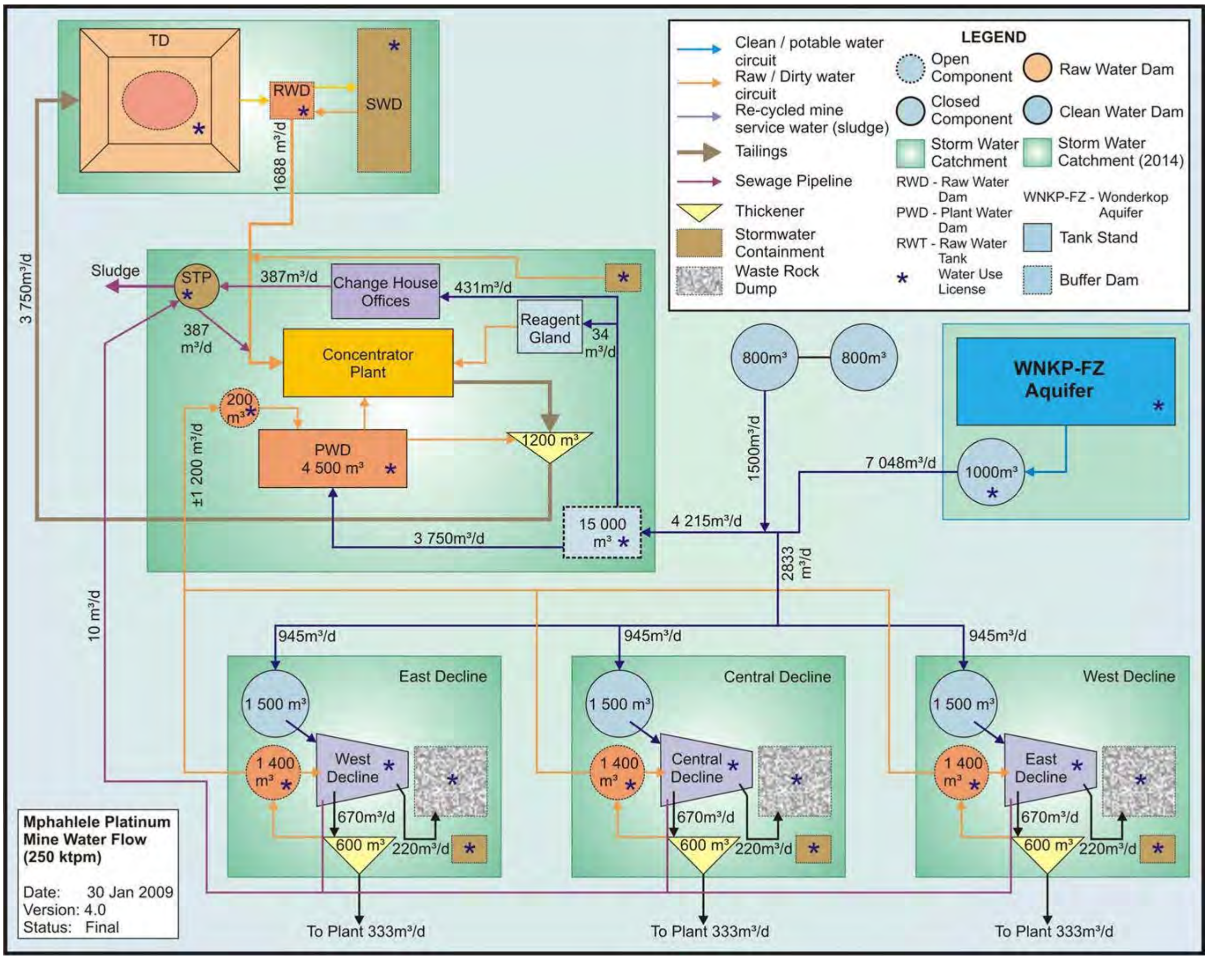


Figure 2-15 Graphic representation of a mine water balance flow system – prior to development (after Vivier, et al. 2009).

The well-known Darcy equation or law that describes flow in porous media is given by (Bear, 1979):

$$q = K\frac{dh}{dl} \qquad (2\text{-}4)$$

Where, q is the Darcy flow (in m/d), K is the hydraulic conductivity and dh/dl is the hydraulic head gradient. Henry Darcy actually used a sandbox model to derive this mathematical model. The groundwater flow equation is derived from Darcy's law and is used to evaluate the potential future behaviour of groundwater head (potential) with time and in space.

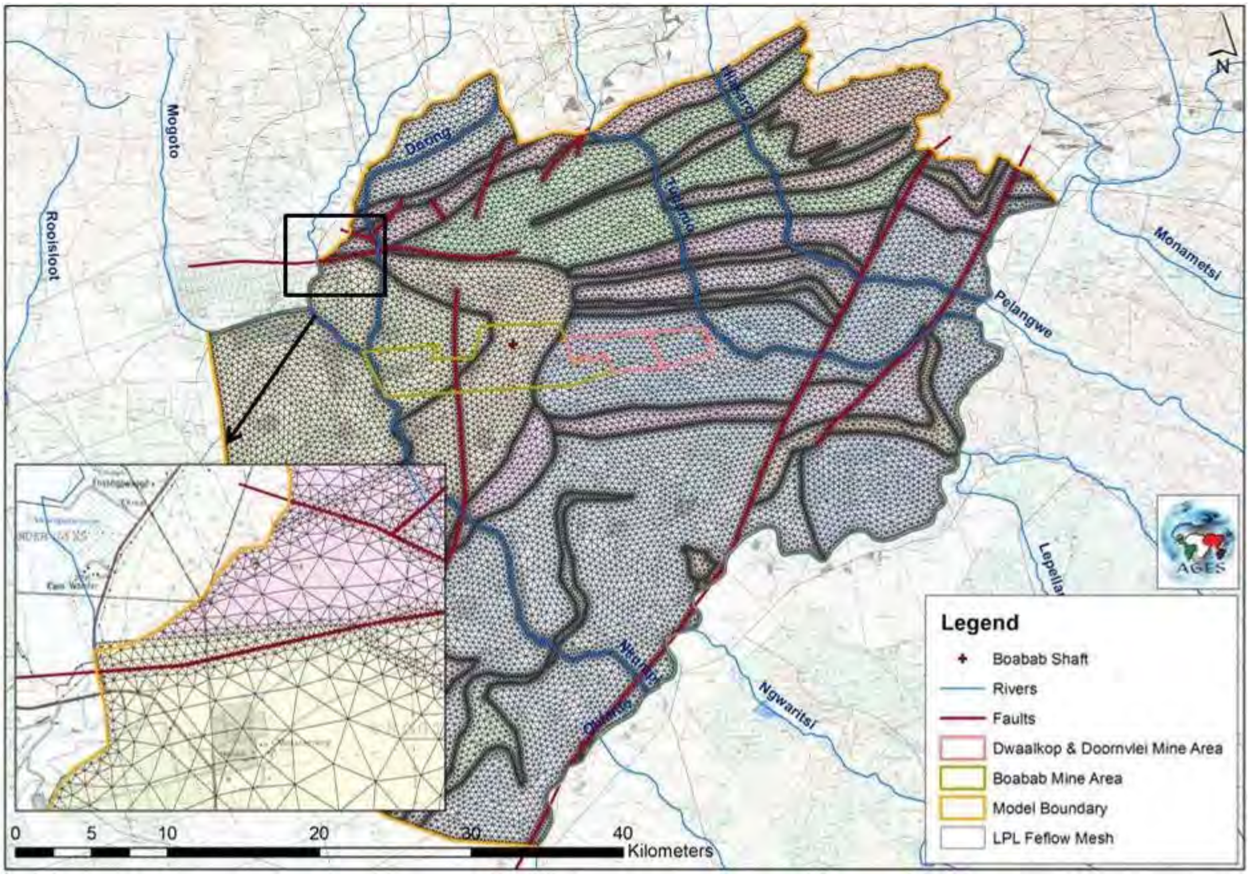


Figure 2-16 Finite element network of a numerical groundwater model (after Vivier, et al. 2009).

The groundwater flow equation for flow in a homogeneous, isotropic and confined aquifer is given by (Bear, 1979):

$$\frac{\partial^2 h}{\partial x^2} + \frac{\partial^2 h}{\partial y^2} + \frac{\partial^2 h}{\partial z^2} = \frac{S}{T}\frac{\partial h}{\partial t} \qquad (2\text{-}5)$$

Where h is the groundwater head elevation, x,y,z, the three-dimensional space coordinates, t, the dimension of time, S the aquifer storativity and T the aquifer transmissivity. Numerical models are all derived from the continuity equation representing the conservation of mass (Bear, 1979; McDonald and Harbaugh, 1988, Spits and Moreno, 1996; Diersch, 2009). Numerical models are developed across the domain of the study area and calibrated against observed field values (Diersch, 2009, Figure 2-16, Figure 2-17).

Groundwater models are used to pre-determine the head distribution over time which is normally used to evaluate the sustainability of groundwater resources or the potential impacts of prospective developments (Figure 2-18). Other uses of groundwater models are; to determine groundwater inflow in underground developments, contaminant transport potential etc.

Quantitative models can either be used as deterministic or probabilistic assessment methodologies to describe physical processes and evaluate potential impacts.

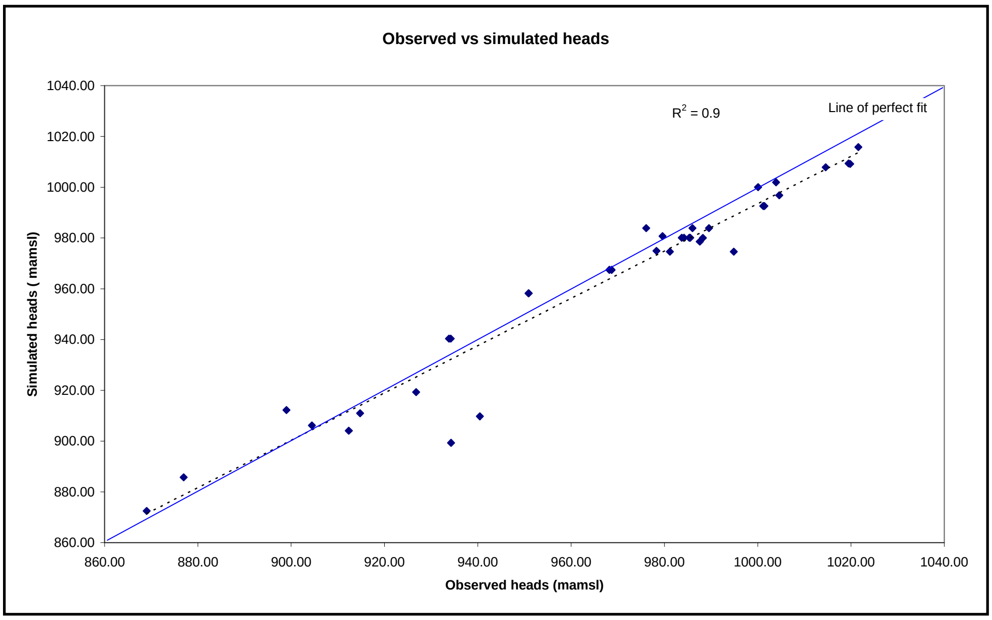


Figure 2-17 Calibrated numerical model with simulated vs observed groundwater heads (after Vivier, et al. 2009).

2.9.8.4 Statistical models

As discussed earlier, there are very few or even none real deterministic physical phenomena. Most observations or test results will have variability in space and time. This is where the use of statistical or probabilistic models becomes important (Basson, et al. 1994). A probabilistic assessment or model of the groundwater flow equation would entail a statistical distribution of the aquifer parameters recharge, transmissivity (T) and/or storativity (S) (Figure 2-19). The model would rerun a large number of simulations (n, typically >100) so that random (or stratified) sampling is done automatically within the sampling distributions. The outcome of heads with time at a specific point or points of interest would also be a distribution.

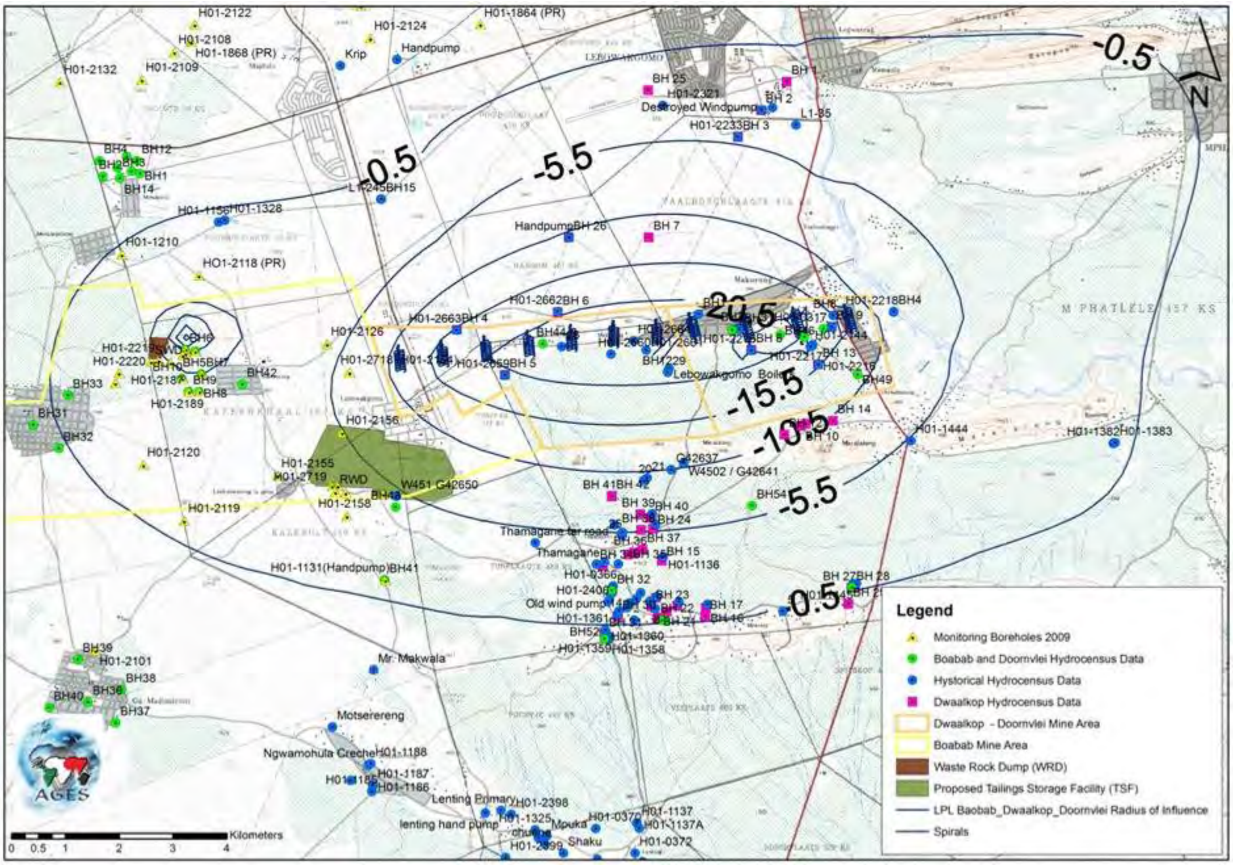


Figure 2-18 Output from a numerical model showing the radius of influence of mine dewatering (after Vivier, et al. 2009).

This modelling method where statistical distributions are sampled in a simulation is used for risk assessments, is known as the Monte Carlo Method (Bear, 1979; Spitz and Moreno, 1996; Wisniewski, 2002). Statistical assessments can be used to determine e.g. a minimum or maximum and an assurance can be related to the outcome (Figure 2-19). It is often used to determine the rainfall at e.g. a 98% level of assurance. The outcome would be to build confidence in that it is possible to determine with an assurance that e.g. more groundwater will be available than the volume quoted (Dakins et al. 1995).

It is thus very useful, but unless the cause of the assessment is justified, has a drawback in that the statistical evaluation of the data and the simulations are time consuming and often impractical.

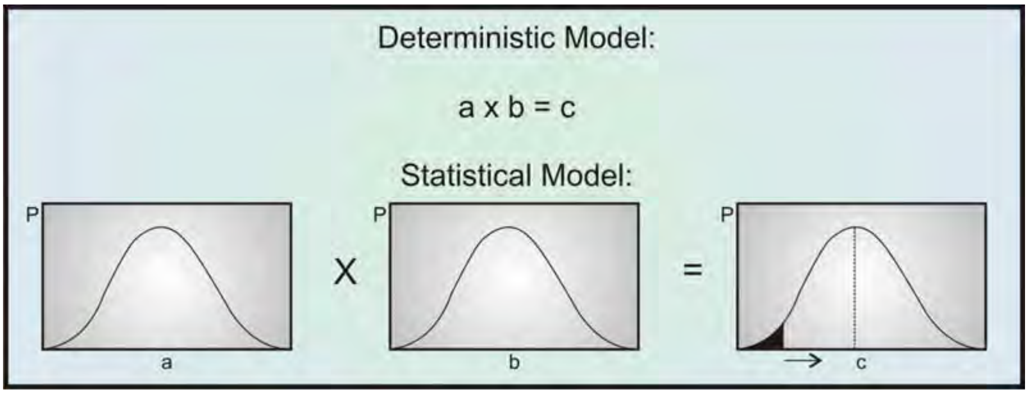


Figure 2-19 Schematic representation of a deterministic and a statistical model.

2.9.8.5 Risk assessments

Statistical evaluations and models are used to determine the probability of an event or outcome. These models can be further used to perform risk[18] assessments where the consequence in terms of e.g. cost or hazard, can be evaluated. Risk assessments can be quantitative or qualitative. Qualitative risk assessments are usually done by ratings done by expert judgment and applied to risk components. Quantitative risk assessments are done to determine the impact on cost or human (and sometimes ecological) health risk (Dakins, et al. 1995).

Safety factor and stochastic risk approaches

In the traditional engineering design approach, a safety factor (F) is defined as (Freeze et al. 1990):

$$F = C/LD \qquad (2\text{-}6)$$

Where C is the capacity and DL is the load of the engineered system. The approach involves designing the capacity (C) based on the projected load (LD) to the extent that it satisfied a predetermined safety factor. It has become more realistic to design safety factors based on a stochastic risk balancing approach where the load and the capacity are represented by probability density functions (PDFs) and the probability of failure (Pf) is defined as the probability that LD

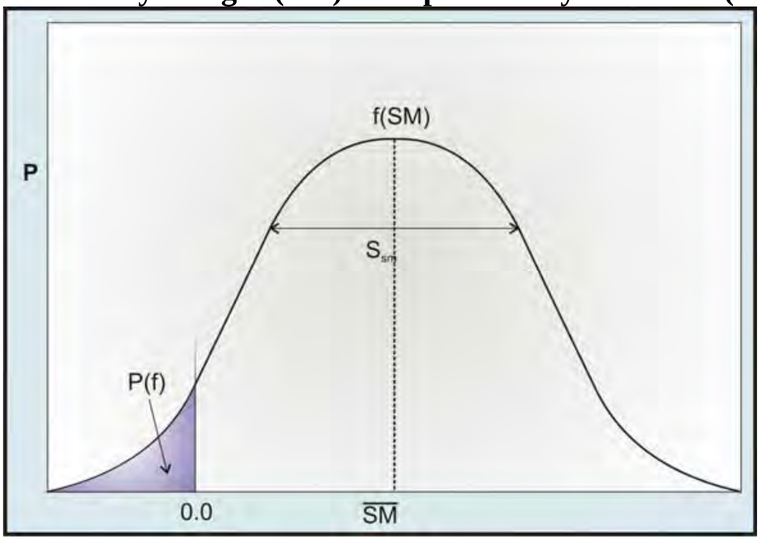
[18] Risk can be defined as the possibility of something bad happening in the future or a situation that could be dangerous or have a bad result (Oxford English Dictionary, 2006). The possibility of incurring misfortune or loss (Collins English Dictionary, 2006). Probability or threat of a damage, injury, liability, loss, or other negative occurrence, caused by external or internal vulnerabilities, and which may be neutralized through pre-mediated action (http://www.businessdictionary.com/definition/risk.html).

exceeds C (Freeze et al. 1990; Mendenhall et al. 2006). A safety margin is defined as the difference between capacity and load:

$$SM = C - LD \qquad (2\text{-}7)$$

The probability of failure is equal to the probability that the safety margin is less than zero. If LD and C is normally distributed with means LD_M and C_M and standard deviations LD_s and C_s , the safety margin (SM) is also normally distributed. In the risk-balancing approach to design, the probability of failure (Pf) is used as the measure of design uncertainty (Figure 2-20).

Figure 2-20 PDF for safety margin (SM) with probability of failure (Pf) (Freeze et al. 1990).



The inverse of the probability of failure (Pf) is the reliability (Rl) given by (Freeze et al. 1990):

$$Rl = 1 - Pf \qquad (2\text{-}8)$$

Reliability is often used instead of the negative connotations related to probability of failure.

Risk-Cost assessments

Risk and cost assessments are done in engineering design to balance the relationship between cost and potential consequence. Risk can be defined as (Watts, 1996; Sandia National Laboratories, 1998):

Measure of Risk = probability of failure x cost or consequence of failure *(2-9)*

The total cost of infra-structure is represented by:

Total cost = Development cost + costs associated with risk *(2-10)*

Risk assessments can be used to make decisions regarding acceptable health exposures to e.g. contaminants in groundwater or budgets that can be allocated to the design of isolation barriers at waste sites or cleanup programmes (Dakins et al. 1995; Sandia National Laboratories, 1998).

If an engineering design has to be done for e.g. a lining at a hazardous waste disposal facility, there are a number of options that can be taken. The risk-averse option would be to aim and design a lining system that can last virtually *forever* (which is not possible). The problem is that to aim and design such a system would be at an infinite cost and even then the risk would not be zero. If an objective function ϕ_j for n alternatives, then the goal is to maximise ϕ_j (Freeze et al. 1990):

$$\phi_j = \sum_{t=0}^{T} \frac{1}{(1+i)} \left[B_j(t) - C_j(t) - R_j(t) \right] \qquad (2\text{-}11)$$

Where ϕ_j is the objective function for alternative j (the cost in R); B_j is the benefits in year t (value in R); C_j is the costs for alternative j in year t (cost in Rand); $R_j(t)$ is the risks of alternative j in year t (cost in R); T is the time period in years and i, is the discount rate as a decimal fraction. The relationship between risk and cost is as such that the risk is never zero no matter how much costs are incurred. The costs increase exponentially with a decrease in risk (Figure 2-21). The reliability increases inversely to the risk, so that there is an optimal engineering design (Freeze et al. 1990; SANRAL, 2006). An absolute risk-averse approach to development would entail very high and impossible costs. This is an important aspect for consideration in sustainable development as economic considerations also form a part of sustainability (Section 2.5; Figure 2-3).

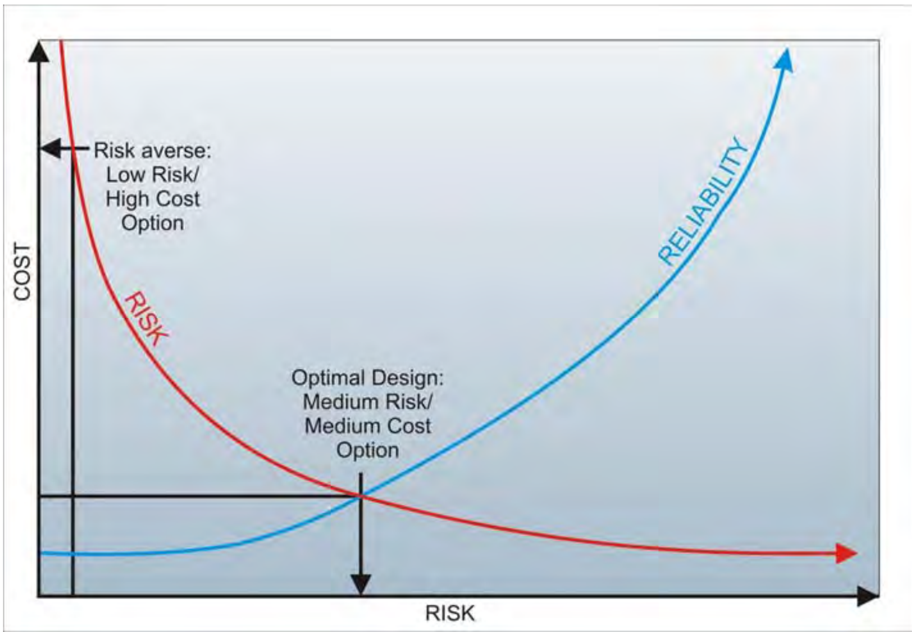


Figure 2-21 Graphic representation of risk vs cost (Freeze et al. 1990).

Health risk assessments

Health risk assessments are important environmental decision variables and are done to determine the potential hazard that e.g. waterborne chemicals or microbes could pose to human health (Watts, 1996; Vivier, 2000). Although health risks are also associated with costs such as medical care, it differs from financial risk assessments in that human lives cannot always be valued financially. There are what is termed as acceptable risk, which would e.g. be in the order of 1.0E-04, meaning that 1 in 10 000 people may die as a consequence of contaminants in water (Watts, 1996).

2.10 Decision-making in environmental management

2.10.1 General

Between 1950 and 2004, the world population increased exponentially from 2.5 billion to 6.4 billion. The world finds itself in the steep part of a growth curve with 6 billion people that could reach the 10 billion mark before 2050 (Figure 2-22; Information box 2-B). Global output, which is a measure of human development, has increased seven fold since 1950. Population growth is therefore the driver behind environmental problems. Natural resources such as minerals, water and waste produced by mines and effluent water that was never thought to be problematic in the 1950's became a challenge before 2000 (Tyler Miller, 2005). This output is directly related to pressure on the environment.

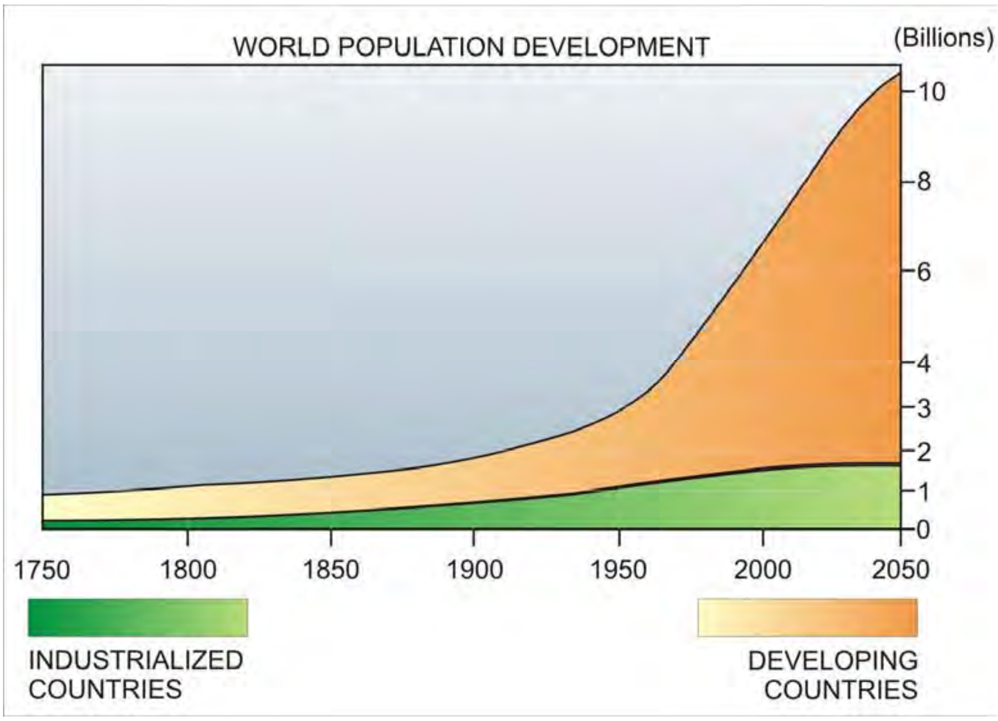


Figure 2-22 World population development graph (www.aylluinitiative.files.wordpress.com).

Environmental management for the purpose of sustainability has never before been a more important aspect influencing the future. Due to the population growth, the major environmental and resource problems that the world is facing today are (Tyler Miller, 2005):

- Waste production
- Water pollution
- Air pollution

- Biodiversity depletion and
- Food supply problems

Although some philosophers maintain that there is no clear definition for *environment* (Strydom and King, 2009), the environment or environmental can be defined as *the natural conditions in which people, animals and plants live* (Oxford English Dictionary, 2006). Another definition is the world we live in, work in and play in, which include all living and non-living things that we encounter on Earth (Aucamp, 2009).

Information box 2-B

Two kings that lived in ancient times had a chess contest and the winner could claim a prize against the loser. After one match, the winner asked the loser to pay him by placing one grain of rice on the first chess square and two grains on the second, four on the third with the number doubling at every square until the 64 squares are filled. The losing king thinking he was getting off lightly agreed with delight. The losing king ended up bankrupting his kingdom because the number of grains of rice he promised was probably more than all the rice that was ever harvested. (Tyler Miller, 2005). At only the tenth square, the value is 1.15E+77!

The legal definition of the environment according to the South African Constitution (Republic of South Africa 1996a) is defined as below. Everyone has the right:

a) To an environment that is not harmful to their health or well-being; and

b) To have the environment protected, for the benefit of present and future generations, through reasonable legislative and other measures that;

    i. Prevent pollution and ecological degradation;

    ii. Promote conservation and;

iii. Secure ecologically sustainable development and use of natural resources while promoting justifiable economic and social development.

The Constitution does not only provide for the natural environment, but also for the other two pillars of sustainability, social and economic development (Section 2.5). The definition of environment and sustainable development in the National Environmental Management Act (NEMA, Act 107 of 1998a) are:

Environment means the surroundings within which humans exist and that are made up of:

i. The land, water and atmosphere of the Earth.

ii. Micro-organisms, plant and animal life.

iii. Any part or combination of (i) and (ii) and the interrelationships among and between them and;

iv. The physical, chemical, aesthetic and cultural properties and conditions of the foregoing that influence human health and wellbeing.

v. Sustainable development means the integration of social, economic and environmental factors into planning, implementation and decision-making so as to ensure that development serves present and future generations.

In this section, the environmental decision-making methods are evaluated in terms of its relation to the three pillars of sustainability.

2.10.2 Origin and characteristics of environmental management

In environmental ethics, the *anthropogenic* view is the perspective that our moral duties regarding the natural world are determined by the duties we owe one another as humans, as well as future generations. Humans are considered as the central component influencing the planet (Glazewski, 2000; http://plato.stanford.edu/entries/ethics-environmental/). The human centred theory[19] has its origin from the Bible in which humans are commanded to subdue the Earth and rule over living creatures (Glazewski, 2000);

"Gen 1:27 So God created man in His own image, in the image and likeness of God He created him; male and female He created them. Gen 1:28 And God blessed them and said to them, Be fruitful, multiply, and fill the earth, and subdue it [using all its vast resources in the service of God and man]; and have dominion over the fish of the sea, the birds of the air, and over every living creature that moves upon the earth." (The Amplified Bible, 1987)

The nature centred theory known as *biocentrism* holds that all living things have an inherent worth by virtue of being members of the Earth's *community of life* (phil-208-environmental-ethics http://ie499.yeralan.org/). Some philosophers maintain that anthropocentrism is limited as it only considers human interest (Glazewski, 2000). Others argue that humans have the biggest influence on the planet and by removing anthropocentrism, it equates to environmental unconcern (de Echeverri, 2007; Manson, 2009). Humans are the only species that can drive themselves to extinction and know that it is or could be happening at the same time. This is a quality developed by an intelligent designer (Information box 2-C; Manson, 2000; Stroebel, 2005). In South Africa, the focus shifted from environmental conservation with the old Environmental Conservation Act (ECA, Act 73 of 1989) to environmental management with the enactment of the National Environmental Management Act (NEMA, Act 107 of 1998). It was realised that any development will have impacts and that development and the environment can or should be managed together. The principles of sustainability require that a balance should be found between the development and the environment (Section 2.5).

Information box 2-C

Suppose we have e.g. wild dogs or humans on an island that have to live off rabbits. If there are only say 150 rabbits on the island and the consumption rate exceed the proliferation rate, then humans would know at some point that they are going to deplete their resource while the wild dogs would just continue until the food source is depleted. The advantage of knowing in advance is that you can do something about it.

Environmental ethics are caught up in a contest or trade-offs between anthropocentrism and biocentrism.

[19] Known as anthropocentricism or humanocentrism.

To be able to e.g. produce meat for humans, cattle must die etc. Environmental management can be equated to chess in a way where the sacrifices that must be made can be limited but not avoided (Section 2.9.6). This is where the concept of *have dominion (or rule) over* in Genesis is important. If it is understood correctly, it would mean that man must also care for nature and not only for himself.

2.10.3 Management and the environment

Management can be defined as getting people together to accomplish desired goals and objectives effectively (using systems) (Hellrieger, et al. 2002). W Edwards Deming developed this management system for the purposes of business process management. He proposed that the business process should be analyzed and measured to identify sources of variations from the original planning that cause products to deviate from the specifications (Hellrieger, et al. 2002; Strydom and King, 2009). The Deming management cycle was developed to describe management and quality control with the Plan-Do-Check-Act (PDCA) process (Figure 2-23; www.balancedscorecard.org; ANSI/ISO/ASQ Q9001-2000, 2000):

1. PLAN: Design or revise the business process and components to improve results or output.
2. DO: Implement the plan and measure or monitor its performance.
3. CHECK: Assess the measurements and report the results to decision-makers. This is the quality monitoring or control component.
4. ACT: Decide on changes needed to improve the process.

Environmental management is not the management of the environment, it is the management of human activities on the environment based on a legal and scientific (decision) process (Strydom and King, 2009). The fact that people are the most important role player in management, distinguishes it from science, where the focus is on nature and physical systems.

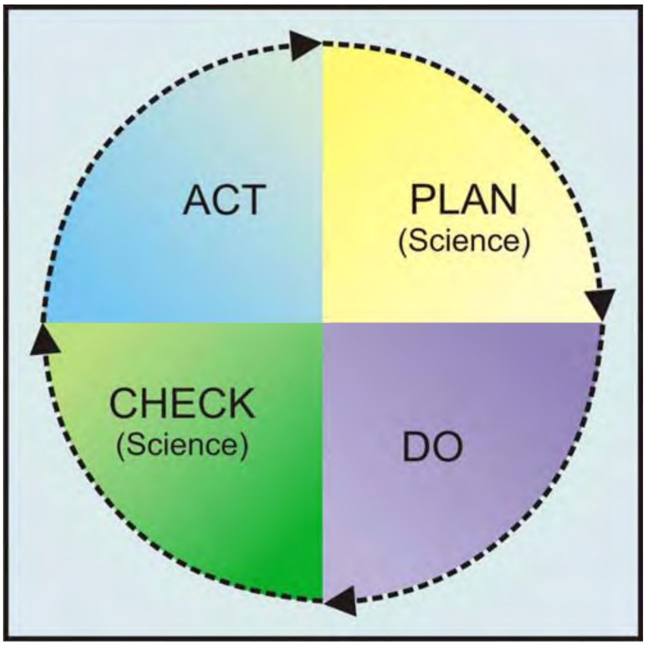


Figure 2-23 The Deming management cycle (www.balancedscorecard.org).

Science is an important input parameter in environmental management (also in general or business management) but management extends further into a domain where decisions have to be made subjectively in the absence of, or with uncertain data and information. Science plays a role in the planning and checking phases where parameters are pre-calculated or where *measuring* is done (Figure 2-23). Science is reductionist and often seeks to differentiate into smaller components and evaluate or analyse in more detail where management seeks to integrate in a holistic way. The scientific and engineering components are included amongst other fields, as specialist studies in environmental management (Table 2-2). Management seeks to arrive at *solutions* and to be able to do that, it needs to integrate different components across boundaries such as science, economics and politics.

2.10.4 Strategic environmental assessments (SEA)

Strategic environmental assessments (SEAs) developed from decision-making theory (Section 2.3, Figure 2-24). In an SEA, a high level approach is followed to identify issues and problems and select solutions based on alternatives (Noble, 2000; Nooteboom, 2000). It is a broad environmental assessment that takes place on a regional scale for planning purposes (Section 2.3.1). An SEA is a process that integrates considerations of sustainability into the formulation, assessment and implementation of policies, plans and programmes (Retief et al. 2007; Aucamp, 2009). Policies, plans, programmes and projects are typically in a tiered relationship cascading down from a broad to a more detail scale (Figure 2-24; Retief et al. 2007).

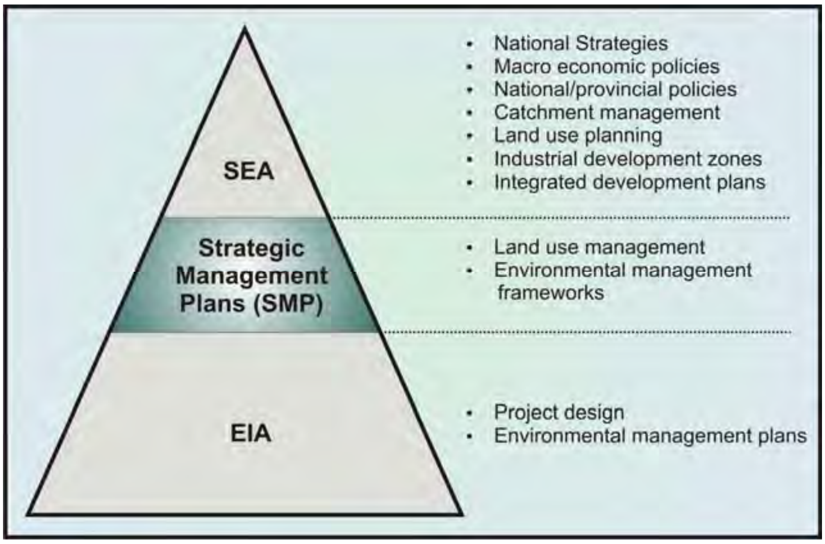


Figure 2-24 Schematic representation of the SEA in relation to the EIA process (Aucamp, 2009).

An SEA can assist to introduce higher level aspects or impacts at an earlier stage of the environmental planning process, before the detailed assessments are made (Harrop and Nixon, 1999). It is intended to compliment the EIA on a pre-level in the process of determining impacts of development activities on a high level. Where an EIA focuses on the impact of specific activities, an SEA proactively assists the decision-maker to determine suitable activities in a regional or specific area before there are any development proposals (Strydom and King, 2009). An SEA is suitable for determining cumulative impacts across a regional area. The advantages of SEAs are (Harrop and Nixon, 1999):

- It encourages the consideration of alternatives that may not be evident on the EIA scale.
- Assists in proactively selecting sites prior to the EIA on a regional level.

- Environmental problems could be anticipated on an earlier stage and solutions are more available on a regional scale.
- Evaluation of cumulative, indirect, synergistic, delayed, regional, trans-boundary or global impacts is more effective.
- The time, effort and cost required for an EIA is reduced or minimised.
- The evaluation of environmental impacts of policies that may not be included in an EIA are made possible.

SEAs are value driven and influenced more by political and legal (Section 2.7 and Section 2.8) rather than technical aspects. On a strategic level, moral issues such as social justice and values (i.e. anthropocentric aspects) supersede technical or natural (i.e. biocentric) aspects (Section 2.10.1; Connelly and Richardson, 2004). The focus of an SEA is based on the broader vision and goals as it is proactive to determine the preferred option/s rather than to determine impacts (Noble, 2000). An example of a strategic environmental assessment is the evaluation of potential environmental impacts of a catchment management plan (Retief, 2007).

SEAs are developed to force the decision-maker not to focus on the detail to be able to distinguish the forest from the trees as in decision-making, there is a trade-off between scale and detail (Section 2.3.1).

2.10.5 The environmental impact assessment (EIA) process

The origin of the environmental impact assessment process started with the enactment of the National Environmental Policy Act (NEPA) in 1970 in the US (Environment Agency, 2000). The EIA process therefore originated as a legal requirement from a political perspective and not from a scientific background (Cashmore, 2004). An EIA is not just a systematic process, it is a structured decision-making framework that acts as an evidence-based instrument that generates information through the use of appropriate techniques (Fisher et al. 2007). The purpose of the EIA process is aimed at finding alternatives and trade-offs in a development of which the *no go* option must be one (Harrop and Nixon, 1999; Aucamp, 2009).

Table 2-2 The EIA process and decision-making (Figure 2-25) (adapted from Aucamp, 2009).

No	EIA step	Purpose/Decision
1	Project definition and motivation	Does the project need an EIA?
2	Scoping (including initial specialist studies)	Identify priority issues on a broad perspective.
3	Specialist studies	Scientific and engineering specialist studies.
4	Impact assessment	Identify potential impacts on the various components of the environment.
5	Draft EMP	Develop environmental management plan for review.
6	EIA Report	Detailed report with environmental management and specialist information.
7	Review	The responsible regulatory authority officially reviews the EIA.
8	Approved / Rejected	A decision is made and a Record of Decision (RoD) is issued by the authority, whether positive or negative. RoD conditions are stipulated based on the management and mitigation measures, which are legal requirements.
9	Monitoring and auditing	The EIA is monitored and audited against the legal requirements.
10	Closure and rehabilitation	All developments have an end of life. Financial provision must be made for closure and rehabilitation towards a post-closure sustainable land use.

Alternatives are environmental options that are listed with the potential impacts and associated management and mitigation measures. The aim of an EIA process is to qualify and if possible to quantify the potential impacts on people and the environment. An important requirement is to identify possible fatal flaws that could result in stopping a development. EIAs make extensive use of Geographical Information Systems (GIS) to determine the spatial coverage of sensitive environs and impacts (Figure 2-26, Figure 2-18). The aim is to identify suitable management and/or mitigation measures that can be implemented to minimise negative effects on the environment. EIAs are characterised by trade-offs in which the significance of the environmental impacts are weighted against the gains of the development (Strydom and King, 2009; Aucamp, 2009).

A particular shortfall that was identified in the EIA process is that in general, too much emphasis is placed on the process and procedures than on the outcomes (Cashmore, 2004).

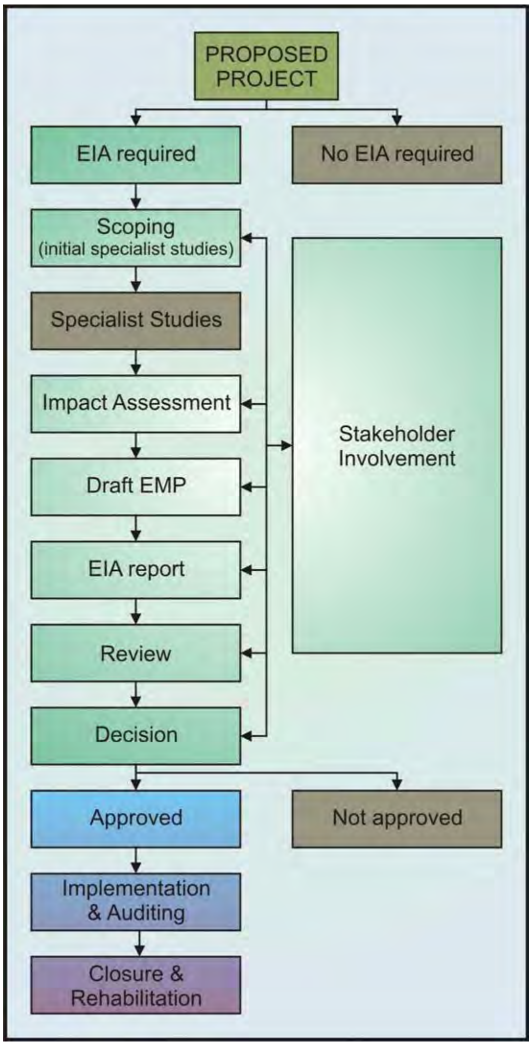


Figure 2-25 Schematic representation of the EIA process in South Africa (adapted from Aucamp, 2009) (Figure 2-23).

2.10.6 Environmental decision-making tools: The impact matrix method

The primary goal of EIAs is to be used as a decision-making tool to assist regulatory authorities to make decisions regarding approval of development projects (Leknes, 2001). All the information from the respective specialist studies is integrated into an impact assessment matrix (IAM) (Table 2-3), which is based on the AHP (Section 2.6.2.1). The impact assessment process is sometimes termed a risk assessment which is summarised in a risk matrix. The two processes are essentially the same with the notation that a formal risk assessment is more quantitative and usually makes use of statistical methods (Section 2.9.8.5). The impact and risk assessments leads to the development of decision-matrices that can be used by developers and authorities to approve or reject projects based on a weighting of both the positive and negative impacts (Leknes, 2001; Aucamp, 2009). The decision matrices in EIAs must be able to take both quantitative and subjective information into account. Impact assessments are done based on (Table 2-3):

1. Project phase, which range from planning, construction, operational to post-operational. Of particular significance are the post-operational and long-term impacts. Specific processes were developed in the radiological safety assessment methodology to identify and rate the post-operational impacts from radioactive waste disposal facilities (Kozak, 1994; Van Blerk, 2000).
2. The activity denotes the specific development action.
3. The probability of the impact, indicates how likely it is to occur.
4. Duration relates to the time that the activity and impact is likely to occur.
5. Scale indicates whether the impact would occur on a large, regional or a small scale on the development footprint.
6. Magnitude or severity indicates the importance of the specific activity and related impact.
7. Significance is used for the final decision-making process. It is listed with and without mitigation measures. The significance is weighted based on the previous six classification criteria.

Table 2-3 An EIA impact matrix for the groundwater component (Figure 2-26)(AGES, 2010a).

Project Phase	Impact Assessment						
	Activity	Probability	Duration	Scale	Magnitude / Severity	Significance	
						WOM	WM

Project Phase	Impact Assessment						
	Activity	Probability	Duration	Scale	Magnitude / Severity	Significance	
						WOM	WM
Construction	Ground water pollution due to storage of chemicals, diesel and building materials	Highly Probable (WOM) Probable (WM)	Short term	Regional	Medium	Moderate	Low
	Oil, grease and diesel spillages from construction vehicles	Highly Probable (WOM) Probable (WM)	Short term	Site	Medium	Low	Negligible
	Spillages from diesel (fuel) facilities	Probable (WOM) Improbable (WM)	Short term	Site	High	Low	Negligible
	Water pollution from inadequate sanitation facilities	Highly Probable (WOM) Improbable (WM)	Short term	Site	Medium	Low	Negligible
Operational	Oil, grease and diesel spillages from mine vehicles during the operation of the plant, opencast extension and underground	Highly Probable (WOM) Probable (WM)	Short term	Site	Medium	Low	Negligible
	Seepage of dirty water into the aquifer during the operation of the plant, opencast extension and underground	Definite (WOM) Probable (WM)	Long-term	Local	High	High	Low
	Pollution of groundwater due to sanitation facilities	Highly Probable (WOM) Improbable (WM)	Long-term	Site	Medium	Moderate	Negligible
	Ground water pollution due to storage of chemicals and mining materials	Highly Probable (WOM) Probable (WM)	Long-term	Site	Medium	Moderate	Low
	Impact on mine water balance – make up water requirements	Highly Probable (WOM) Probable (WM)	Long-term	Site	Medium	Moderate	Low
	Spillages from diesel (fuel storage) facilities	Highly Probable (WOM) Probable (WM)	Long-term	Regional	High	Moderate	Low

WOM = Without mitigation measures. WM = With mitigation measures.

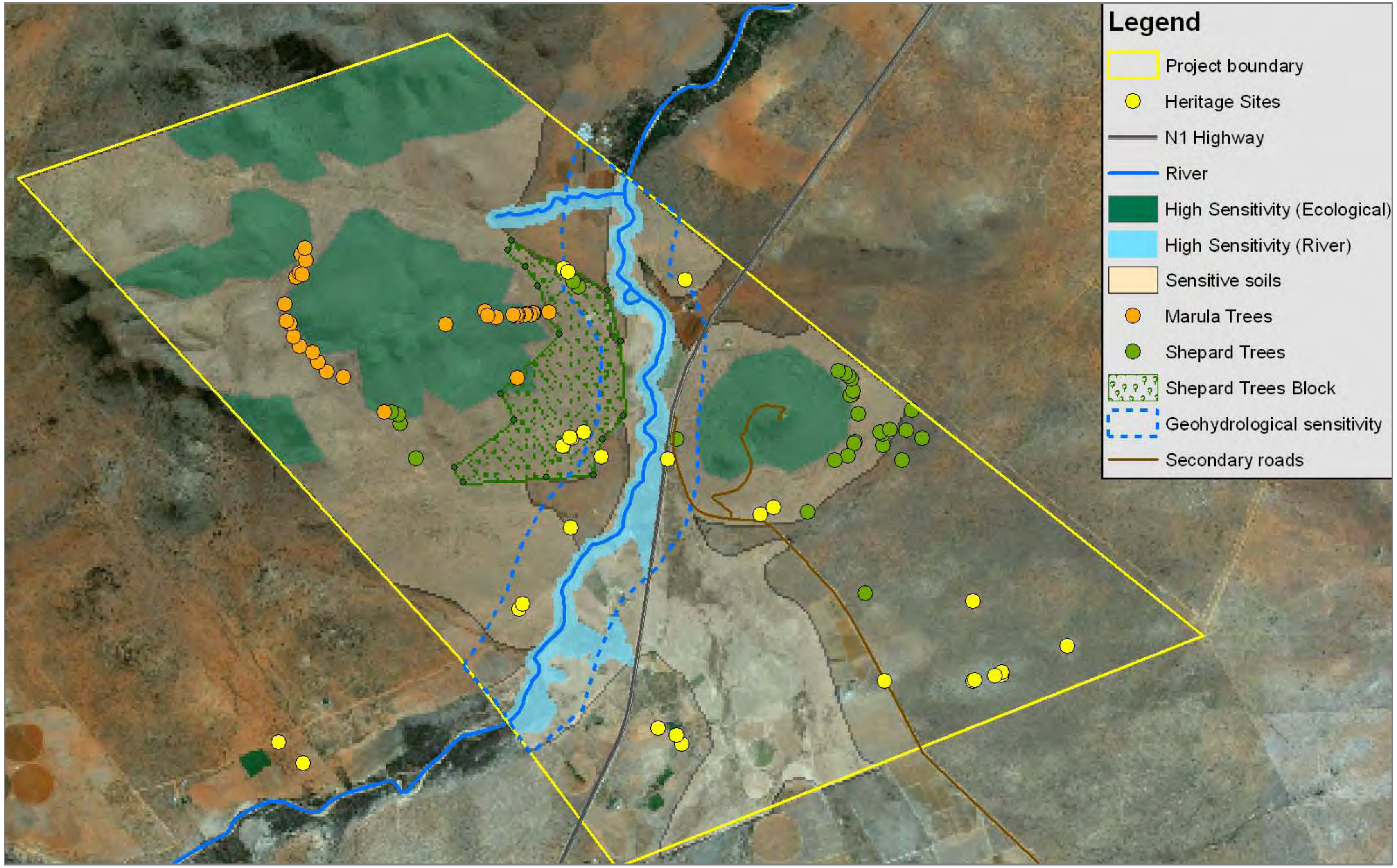


Figure 2-26 An EIA spatial environmental sensitivity map (AGES, 2010a).

An important outcome of the impact or decision matrix is that it must qualify the significance of impacts (DEAT, 2002; DEAT, 2004). The rating or weighting of the severity of activities is based on scaling, weighting and aggregation (DEAT, 2002). The purpose is to provide a reference base that provides the information for decision-making. A development that has activities with high probability, long-duration, large scale and a high severity, even with mitigation measures would weigh towards a rejection. A development with activities that has a high probability but short duration on a small scale and a low severity and hence low significance would weigh towards a positive outcome or acceptance of the development. The problem is that most decisions are based on contradictions between positive and negative impacts that makes it difficult for the decision-makers.

2.10.7 Environmental decision-making and uncertainty

Most natural processes and therefore also EIAs are associated with uncertainty. Uncertainties influence the reliability in impact forecasts, but can be reduced with specialist studies and research. The main causes of uncertainty in environmental assessments are (Sager, 2001; Aucamp, 2009):

1. Scientific uncertainty that is inherent in natural processes and where there is a limited understanding of a natural or social system or of a process that drives change.
2. Data uncertainty is introduced by inadequate or non-comparable data. This includes sampling and analysis variation.
3. Policy uncertainty occurs where objectives are unclear or disputed or where standards and guidelines are unclear.
4. Reliability and accuracy of information could take place where higher precision could lead to lower accuracy since errors could propagate.
5. Relative importance of different types of impacts: Impacts on humans or human health are considered of a high importance even when associated with high levels of uncertainty.
6. Variability of the natural environment such as flood or drought recurrences.

7. Inherent imprecision of models such as physical or mathematic models where a deviation from the actual system is introduced through simplification or assumptions.

Uncertainties are addressed through scenario development and scenario modelling of best and worst cases. Statistical analyses of data are done using methods that caters for characterization of uncertainty. Sensitivity analyses to determine the potential influence and variability of parameters are useful. It is used to evaluate the influence of uncertain parameters on impacts. The introduction of e.g. the precautionary principle (NEMA, Act 107 of 1997) to determine the conservative cases for specific impacts are used to eliminate the effects of uncertainties (Section 2.7.4).

2.10.8 Environmental economics

Economics is an integral part of sustainable development as it has its basis on how to deal with scarce resources (http://ec.europa.eu/environment/enveco/index.htm). The role of any monetary currency is to be able to make comparisons[20]. To state the magnitude of environmental impacts have little meaning in planning or project management if no economic value can be linked to the impacts or at least to the mitigation of the impacts (Hecht et al. 1999; DEAT, 2004; Strydom and King, 2009). It is accepted that not all environmental impacts can be determined and costed analytically. The costs of biodiversity loss or climate change are not well understood. It was e.g. found that urban property prices decrease away from forest areas (Tyrväinena and Miettinen, 2000). Environmental significant areas and water bodies such as the Hartebeespoort Dam attract upmarket residential development. The environment has intrinsic value e.g. intact wetlands slow floodwaters, filters contaminants and recharges aquifers (Parsons, 2006). Environmental and resource economics uses a differentiation of values defined by the *total economic value* (TEV) to determine environmental worth (Hecht et al. 1999). The total economic value of an ecosystem consists of the following uses; *direct use*, *indirect use*, *option to use* and *non-use values*. Direct use could be generated by the consumptive or non-consumptive use of a resource. Indirect value could form inputs to production elsewhere in the economy or by saving of costs. Non-use

[20] If one were to say that “I bought a vehicle”, then it is not clear what type or class of vehicle was bought. If one were to say that “I bought a vehicle for R 1 million, then there is an immediate understanding of what type of vehicle or class it could be.

provides value in relation of the option for future use (Strydom and King, 2009). The bequest value is the willingness to pay for future generations to see e.g. endangered wildlife like the cheetah. The TEV can be presented by (Hecht et al. 1999);

$$\text{TEV} = \text{use value} + \text{non-use value} \quad (2\text{-}12)$$

and

$$\text{Use value} = \text{direct use value} + \text{indirect use value} \quad (2\text{-}13)$$

with

$$\text{Non-use value} = \text{existence value} + \text{option value} + \text{quasi-option value} + \text{bequest value} \quad (2\text{-}14)$$

An example of the components of the total economic value for a rainforest is shown in Table 2-4.

Table 2-4 Total Economic Values for a Tropical Rainforest (Hecht et al. 1999).

Direct uses	Indirect uses	Option Values	Existence Values
Timber Wild plants Hunted animals Medicines	Recreation Wind protection Air pollution control Watershed protection Nutrient cycling Carbon fixing	Future direct and indirect uses	Willingness of westerners to support protection of tropical forests, e.g. based on moral conviction

The principle should be that if say a new mine or development is planned, but the revenue that it could generate is less than the cost of environmental rehabilitation in the closure and post-closure period, then it should not be opened. Funds should be allocated or provided to any project during the operational phase to account for environmental costs (MPRDA, Act 28 of 2002, Figure 2-27; Section 2.6.3).

The economic viability is an important component of environmental management in line with the triple bottom line (Section 2.5; Figure 2-3). Economic valuation methods that are used in environmental management include the NPV (or NPC) method and the CBA methods (Section

2.6.3). These methods are also used in trade off assessments that are valuable in environmental decision-making.

2.10.9 The role of technology in sustainable development

We are living in a time where the speed of technological development is exponential and the technological landscape is ever changing (Dormann and Holliday, 2009). Technological development was/is responsible for the rapid degradation of the environment since the industrial revolution (Tyler Miller, 2005). Technological development could either harm or improve the environment with developments such as more effective (i.e. faster) mining and farming methods that degrade the environment at a much higher pace. On the other hand clean development mechanisms (CDMs) such as wind energy or fuel cells reduce the dependence on fossil fuels and lower environmental impacts. The development of recycling, water saving and water treatment technologies can be used to remediate degraded natural environments (Tyler Miller, 2005).

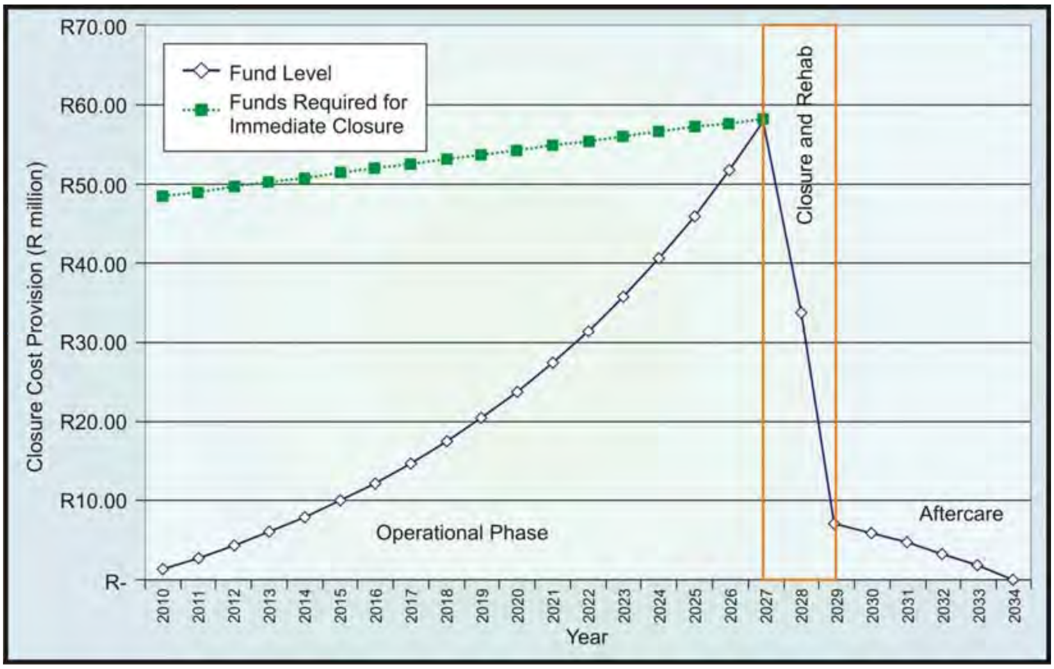


Figure 2-27 Environmental economical evaluation of a new mine development (AGES, 2010a).

Apart from more responsible use of resources, the development of environmentally friendly technologies is one of the hopes for human and natural sustainability. A problem is the barrier to market entry, as sustainable technologies are usually more expensive than existing technologies with well established infra-structure (Balachandra et al. 2009). From the CDMs, wind energy and hydro power technologies are the furthest developed along the market entry curve with hydrogen fuel cells and geothermal technology the least developed (Figure 2-28).

Although major positive changes and environmental initiatives such as carbon credits have evolved, there is still a lot that must be done to integrate the environmental technologies on the policy and political decision-making level (Sheate and Partidário, 2009, Patterson, 2010). The market entry of CDMs will be slow without political and policy intervention to generate economic incentives that could fast track the technological development and implementation.

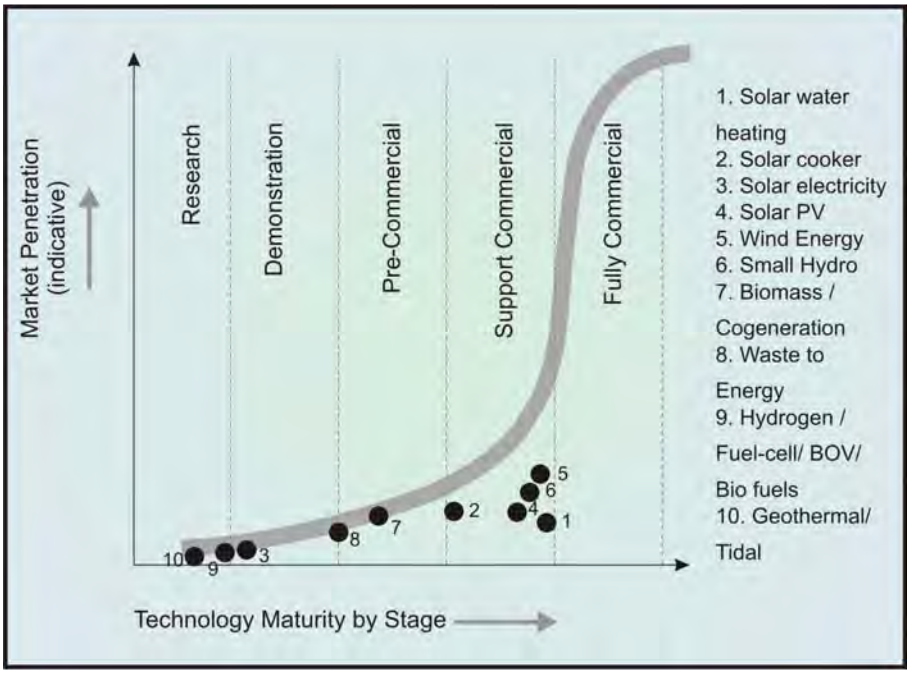


Figure 2-28 Position of CDMs in pursuit of commercialization (Balachandra et al. 2009).

2.11 Summary

Decision-making is an important part of human existence. It is the most important aspect that faces governments and individuals on a daily basis. It is how effective decisions are made that determines the outcome of the future, whether positive or negative. The making of a decision entails a cognitive process that involves the evaluation of options and the choosing of one or more of the options or a sequence of options. This could be done by following a formal process in a business or scientific study or by making a judgement in a split second in an accident scenario.

Decision-making in the context of sustainability requires the integration of the components as contained in the triple bottom line (Table 2-1, Figure 2-3). It is not possible to generate effective decisions in terms of sustainability by considering aspects from a social, technological or environmental perspective in isolation. The consideration of decision-making methods and principles within the social (business & economics, legal and political), science and engineering and environmental fields must be considered holistically. Some of these methods require quantitative, some qualitative and some completely subjective approaches (Figure 2-29). The different disciplines that vary from quantitative to subjective have to be integrated to qualify for sustainability. Failure to do it creates an *invisible but opposite pressure* that opposes integration and hence sustainability (Figure 2-4).

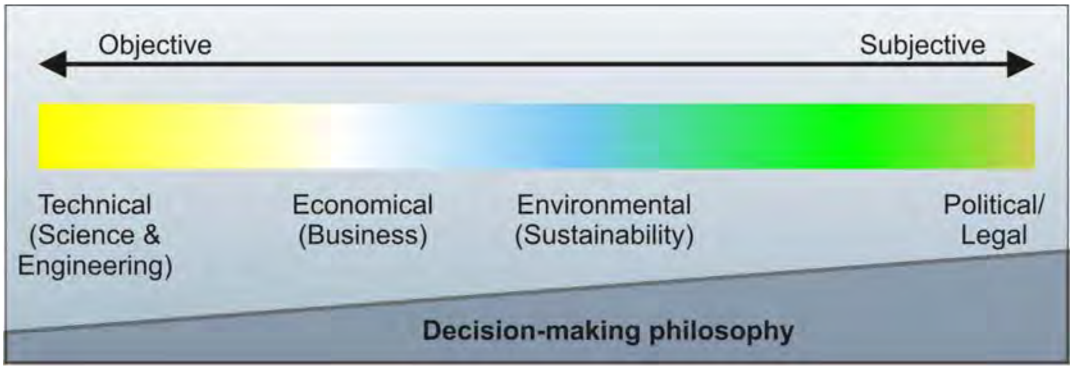


Figure 2-29 Subjective to objective range in the decision-making domain (own construction).

The philosophy and principles of the decision-making processes in the various disciplines were evaluated to determine differences and similarities and how the differences can be overcome and the methods be utilised for the purposes of sustainability (Table 2-5). No decision-making process can be followed without a philosophy that determines the perspective of the process (Figure 2-29). The philosophy of a country, province or developer will determine to what extent sustainability can be reached. If a poor nation is in need of food, the wildlife sanctuaries will be used to obtain food. Rich nations would be reluctant to e.g. sign the Kyoto Protocol to prevent environmentally friendly, but more expensive energy sources etc. The pillars of sustainability rest on the pillars of environmental, social (political, legal and economical) and technical (development), which in turn is founded upon a philosophy that serves as the basis for actions (Figure 2-30).

In the business and economic sphere, decision-making is done through formal processes. The rational model requires a thorough process of decision goals and objectives defined, followed by identification of alternatives, choosing an alternative, with implementation and follow up and control. The problem with the rational method is that it could become too exhaustive to evaluate all the options available. It was found that most business decision-makers follow a bounded rationality model. The bounded rationality model represents the decision-maker’s inclination to select less than the best goal or alternative, to engage in a limited search for alternative solutions. In this model, it is accepted that information and control over external and internal environmental aspects influencing decisions will be inadequate. It is then practical to select an acceptable goal or alternative rather than to spend time and money on an elaborate research for the best or optimal alternative. This model is suitable for decision-making using sparse data and is characterised by *limited search*. Limited search is where a detailed search is not done, but the tendency is to consider the options until an adequate one are found. The limitations associated with practical environs with time and money constraints, favours the limited search option. Most decision-makers therefore satisfice rather than optimize.

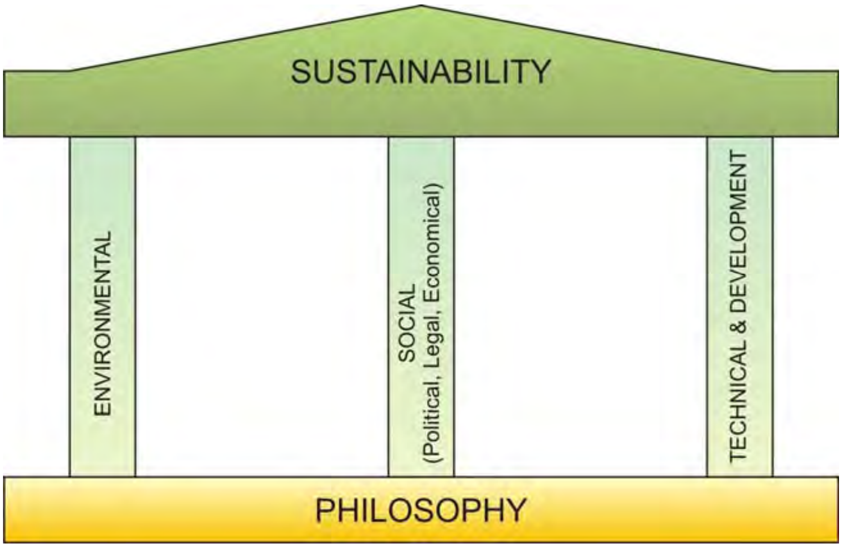


Figure 2-30 The building of sustainability based on the three pillars founded on a philosophy (own construction).

Quantitative methods in business decision-making are the NPV (or NPC where costs are involved) and CBA methods that account for future discounted cash flows and a comparison of options and costs.

Decision-making in the legal sphere is important as environmental regulation is done using legal processes. Water management in South Africa is regulated by legal processes such as the National Water Act (NWA; Act 36 of 1998) and the National Environmental Management Act (NEMA) (Act 107 of 1998). Any process should take into account that if disputes arise, it will be done by judicial processes. Technical processes often encounter problems when challenged in this way. In judicial processes, there is a distinction between criminal and civil cases. In general, criminal cases require proof beyond reasonable doubt and in civil cases, there is a gradation of proof and guilt can be split between two or more parties. There is an argument that judicial proof is different from mathematical proof, but that mathematical proof could be very useful in judicial proof.

Table 2-5 Comparison of environmental decision-making components in business, judicial, political, science & engineering and environmental management (own construction).

No	Triple bottom line	Decision-making field	Decision methods	Quantitative	Qualitative	Subjective
1	Social	Business (economics)	Rational method	M	L	VL
			Bounded rationality method	M	M	VL
			Quantitative methods	H	L	N
		Judicial (legal)	Rule and value driven decision-making	L	M	H
		Political	Power driven political method	N	L	H
2	Technology (development)	Science & engineering (technology)	Quantitative methods	H	M	VL
			Deterministic	H	N	N
			Statistical	H	M	N
			Assumptions	VL	M	H
3	Environmental	Environment	All of the above	H	H	H

Legend	
High	H
Medium	M
Low	L
Very low	VL
None	N

Although mathematical probabilities could play a role, judicial processes follow principles and values rather than mathematical probabilities. The following are important judicial decision principles:

- Innocence until proven guilty. Any accused in a criminal case will be presumed innocent until proven guilty.
- Proof beyond reasonable doubt for criminal cases and balance of probabilities for civil cases. Causal effects between the accused and actual damages must be proven.
- Interrogation and cross-examination of data, information and conclusions by both sides of the argument to determine what qualifies as evidence.
- A case is built for or against an argument and the case that weighs the most usually wins.
- Burden of proof is on the accuser. The onus is on the accuser to prove that the accused is guilty.

- Polluter pays principle. If found guilty, the polluter must be held liable for financial loss or costs incurred. These could involve future costs or lost opportunity for income.
- Precautionary principle. Where decisions that could involve potential environmental impacts should be made in favour of or conservatively to protect the environment.

Another specific gap exists between science and judicial processes where scientific theories, parameters or assumptions are viewed as truth. The gap between the legal and technical processes in environmental management could cause problems for sustainable development. The technical or legal processes could contradict each other which would mean a much more difficult road to sustainability. Although scientific probabilities might be used in legal cases, the merit based on values would be more influential.

Environmental and sustainable development receives focus in the global arena and is in cases highly politicised. Political motives are well developed in South Africa and it can strongly influence or even override all the other disciplines. The influence that politics have on environmental decision-making is critical and should be accounted for in sustainability assessments. The political model for decision-making represents the self-interests and goals of powerful external and internal stake holders. Power is an important aspect and represents the ability to influence or control individual, team, departmental and organizational decision-making and motives. Political processes are most likely to be utilised when powerful stakeholders and decision-makers disagree over the most important goals, especially when analysts (technical, business or environmental) do not succeed in reaching acceptable or suitable alternatives. Political intervention can either strengthen or hamper sustainable development. In most cases, it aims to please the majority of the people and not the basis of the cause.

Science consists of hypotheses, theories and experimental proof. When a theory has been tested and accepted it can also be termed a law. Scientific discoveries often start with an idea, which is formulated into a hypothesis that is accepted as a theory when it is experimentally proved or universally accepted. The scientific method assists to organize thoughts and procedures in an organized way. Scientists make use of thoughts and/or observations that lead to the formulation of hypotheses and deductions to reach conclusions. An important component for science is to make the link between cause and effect through experimentation or calculation.

Logically, there is a fundamental distinction between two forms of scientific reasoning known as deductive and inductive reasoning. If the information that is relied on in good deductive reasoning is true, then the conclusion is guaranteed to be true. In the case of good inductive reasoning, the truth of the conclusion is not guaranteed, but based on its premises the truth is likely. Deductive reasoning leads to deterministic and inductive reasoning to probabilistic outcomes. Science is therefore defined more by its method than by its subject matter. The notion that there is something like an exact science is therefore not true. Science is not exact, but stands on the most probable outcome based on experience or probable theories. Scientists can therefore do two things, they can disprove things and it can be determined that a particular model has a high probability of being true. Scientifically, no theory can be proven as absolutely true.

Statistical methods for analysis have proven to be powerful decision-making tools. Bayesian statistics accepts that statistical variation such as the mean, median, standard etc can be inferred based on known information or a prior judgement. In probability theory, the Bayes Theorem indicates how one conditional probability is dependent on its inverse. The use of the Bayesian approach to statistical analysis is useful as it is suitable for use where data is sparse or limited. The Bayesian approach involves initiating the evaluation using limited or even analogue data, which is re-evaluated iteratively as actual and more data becomes available until the process converges. It is ideally suited for decision-making, especially for the purposes of sustainability, where data is often sparse or does not exist. Scientific methods such as the statistical decision tree, optimization, game theory, conceptual and mathematical models are important decision-making tools.

A scientific model can be defined as a conceptual, dimensional or mathematical simplified representation that acts as an analogue of the actual or real world system. Scientific models usually represent relationships between variables. The purpose of a model is to pre-determine an outcome of a natural (or economical) process or event before it actually happens using prospective evaluations. Models are in essence decision-making tools that should assist the analyst in understanding the problem. Models have become the principle tools of modern science that assist the analyst to *see beyond the horizon*. Models generate some of the biggest controversy in science and engineering. This led to the well-known quote by George Box that *All models are wrong, but some are useful*. The use of models forces the analyst to integrate all the data and think about the problem. It assists in natural resource quantification and understanding and is a decision-making tool that is growing in importance. Models are used in environmental impact studies to determine potential spatial and temporal impacts on the environment. Statistical models are used for risk-cost assessments, especially where engineering designs or decisions are involved.

Environmental management is not the management of the environment. It is the management of human activities on the environment based on a legal and scientific decision-making process. The fact that people are the most important role player in management, distinguishes it from science (where the focus is on nature). Science is an important input parameter in environmental management (also in general or business management) but management extends further into a domain where decisions have to be made in the absence of, or with uncertain information. Science often seeks to differentiate into smaller components and evaluate or analyse in more detail where management seeks to integrate. Science plays a role in the planning and checking phases where parameters are pre-calculated or where measuring is done.

Strategic environmental assessments (SEAs) developed from decision-making theory. In an SEA, a vision is developed to identify issues and problems, assemble issues and viewpoints to determine alternative solutions and select courses of action. It is a high level environmental assessment that takes place on a regional scale for planning purposes. The EIA process originated as a legal requirement from a political perspective and not from a scientific background. An EIA is not just a systematic process, it is a structured decision-making framework that acts as an evidence-based instrument that generates information through the use of appropriate techniques. The purpose of the EIA process is aimed at finding alternatives and trade-offs in a planned development. Alternatives are environmental options that are listed with the potential impacts with management and mitigation measures. The aim of an EIA process is to qualify (if possible to quantify) the potential impacts on people and the environment. Decision-making methods that are used in EIAs are the impact or decision-matrix and spatial sensitivity maps. Environmental decision-making for the purposes of sustainability is broad and encompasses quantitative, qualitative and subjective methods.

Environmental and resource economics uses a differentiation of values defined by the total economic value (TEV). The total economic value of an ecosystem consists of direct use, indirect use, option to use and non-use values. The economic viability is an important component of environmental management in line with the triple bottom line. Economic valuation methods that are used in environmental management include NPV, NPC and CBA methods. These methods are also used in trade off assessments that are valuable in environmental decision-making.

Technological development is responsible for the rapid degradation of the environment during the industrial revolution. Technological development could either harm or improve the environment. Developments such as more effective (i.e. faster) mining and farming methods could degrade the environment at a higher pace. On the other hand CDMs such as wind energy or fuel cells reduce the dependence on fossil fuels. The development of recycling, water saving and water treatment technologies can be used to remediate degraded natural environments and future technological developments could bring hope for sustainable development.

2.12 Discussion

Decision-making is the process of choosing the best of alternative options or to develop new options if the existing alternatives are inadequate. The founding philosophy is considered as the most important basis of decision-making. There are often contradictions in sustainable development assessments that have to be dealt with from a philosophical context as quantitative and qualitative approaches cannot interpret contradictions[21]. Contradictions could occur in technical data or assessments that have to be dealt with from a philosophical perspective that would seek a balance between contradictions or choose a value-based outcome.

In decision-making, there are three basic errors that can be made. The first is by operating only from a gut feel or intuition and not analysing the options that leads to oversimplification of the process and not choosing the optimal (or right) options. The second is an over-analysis of the situation that leads to violation of time and cost constraints. The third is by following a silo approach by over-considering one discipline and neglecting others. This happens mostly because the analyst is trained in a specific discipline and may not have sufficient knowledge or a holistic[22] view of the problem.

[21] An example of a philosophical contradiction is:" Luk 13:30 (Luk 13:30) And behold there are last which shall be first, and there are first which shall be last. (Holy Bible, King James Version).

[22] Holism (from the Greek word *holos* meaning entire, all or whole) is the philosophy that the whole of something must be considered in order to understand the different parts (Oxford English Dictionary, 2006). Practically it means that the whole is more than the sum of its parts. Reductionism is sometimes seen as the opposite of holism. Reductionism in science says that a complex system can be

Decision-making can be done on various levels. Strategic decision-making takes place on a broad base without the consideration of too much detail. This is usually followed by tactical decision-making that would consist of an implementation plan of the strategy, typically containing actions and time frames. Detail can obscure the issues of strategic importance, where the proverbial *can't see the forest from the trees*[23] problem arises. Hence the importance of a broad based decision-making approach or strategy before the detail is considered and not vice versa. Tactical decision-making is on-going and is used to implement and monitor the overall strategy.

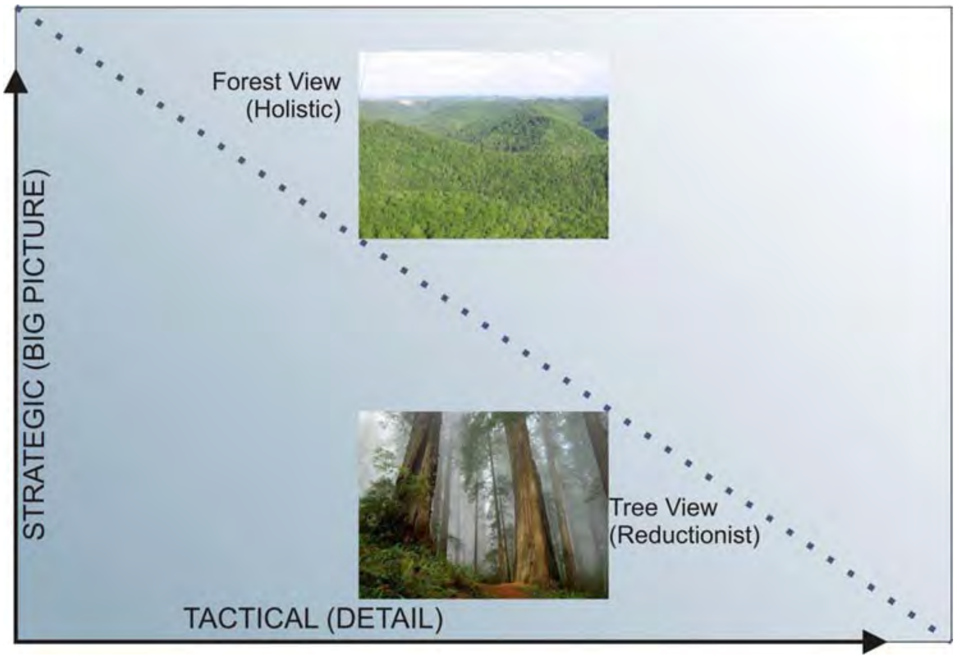


Figure 2-31 Schematic representation showing the difference between detail and broad based decision domains (own construction).

There is a contradictory aspect between strategic or high level and tactical decisions where detail are required (Gasper and Clore, 2002; Förster and Higgins, 2005).

explained by reduction to its fundamental parts (Gorham, 2009).

[23] Proverb meaning overly concerned with detail; not understanding the whole situation. Used when expressing that a person is focusing too much on specific problems and is missing the point (http://esl.about.com/ library/ glossary/bldef_130.htm).

The broad base and detail cannot be evaluated at the same time by the same person. It would be like trying to focus at two different points at the same time, which is an impossible action. Broad based or holistic decision-making is primarily a right brain (i.e. holistic) and detailed analytics a left brain function (i.e. reductionist) from a neurological perspective (http:// performance-rules.com/ performance/ right-brain-thinking -in-a-left-brain-world).

In management, a holistic approach is followed whereas in science a reductionist approach is followed (Figure 2-31; Footnote 22). Decision-making requires an integration of science for measurement and management to get people to plan and act (Figure 2-32). A holistic approach should be followed first to see the forest boundaries and not focus on the trees. The purpose is to firstly determine the problem boundaries (i.e. frame the decision problem) before the internal variables are considered. Once a strategic decision has been taken, it should be followed by tactical (i.e. detailed and reductionist) decision-making, where the detail is considered in a gradational and iterative approach (Figure 2-31).

By starting with the detail or only stopping on the high level would compromise the quality of the decision-making process and the outcome. Research has proved that the success of decisions is positively correlated if following a formal decision-making process (Dean and Sharfman, 1996), especially if a holistic approach is followed first (Eigenbrode et al. 2007). A formal decision-making process that makes use of the relevant decision-making tools assists the analyst to *see beyond the horizon* and enable the making of decisions in more complicated problems.

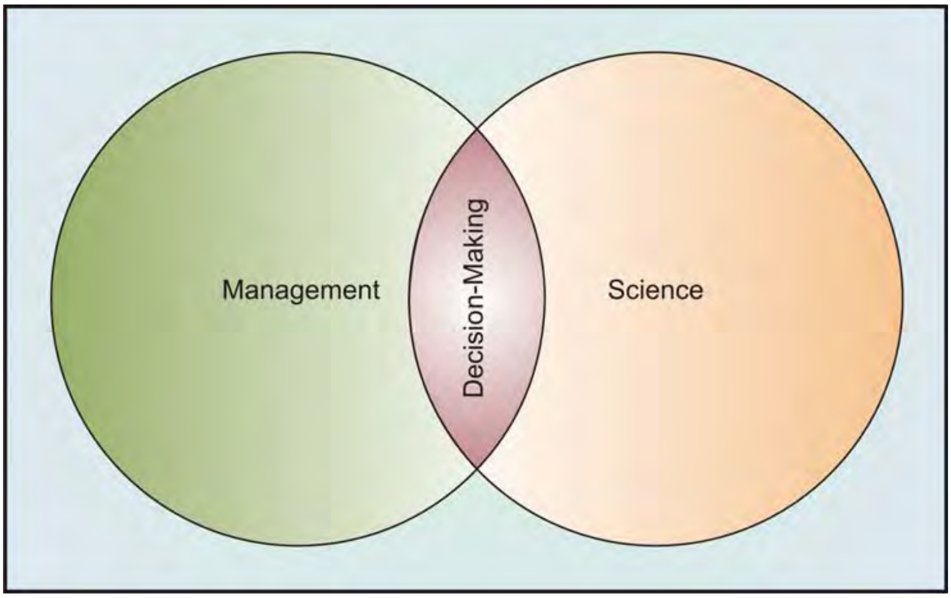


Figure 2-32 Decision-making shown graphically as the intersection between science and management (own construction).

From this assessment, it is evident that sustainable development is a process that behaves like a system. A system acts like a chain that is only as strong as its weakest link. The principle of sustainability requires that decision-making methods and constraints must be integrated in a holistic manner to prevent *unbalanced outcomes* (Table 2-5*)*. Unbalanced outcomes are when the best political or financial decisions are made in isolation.

CHAPTER 3

"What do we live for; if it is not to make life less difficult for each other?" TS Eliot

3 DATA AND INFORMATION IN THE DECISION-MAKING PROCESS

3.1 Introduction

The basis of any decision process whether quantitative or subjective is *information*. Information is based on *data*[24] and the quantity and quality of the data will influence the quality of information. Data and information are therefore the building blocks of the decision-making process. It is important to consider the role or influence that data and information could have on the decision-making process. The following questions often influence environmental decision-making:

1. Does more data equate to better information?
2. What is the ideal state of data and information in decision-making?

In this chapter, the role of data and information in the decision-making process is evaluated to characterise the process. It is evaluated whether more data is better through an application on groundwater as an environmental component.

3.2 Information Science

There are a number of scientific and theoretical approaches that study information and aim to define it. Information was the object of philosophy even before the information age involving computers. A number of attempts to define the concept of information failed to produce a single, unified definition. The concept of information is a polymorphic phenomenon and a polysemantic concept (Stanford Encyclopaedia of Philosophy, 2010).

[24] In general, raw data that (1) has been verified to be accurate and timely, (2) is specific and organized for a purpose, (3) is presented within a context that gives it meaning and relevance, and which (4) leads to increase in understanding and decrease in uncertainty. The value of information lies solely in its ability to affect a behaviour, decision, or outcome. A piece of information is considered valueless if, after receiving it, things remain unchanged (www.businessdictionary.com).

From the various concepts, an information map can be produced that shows the information types and processes of information flow (Figure 3-1).

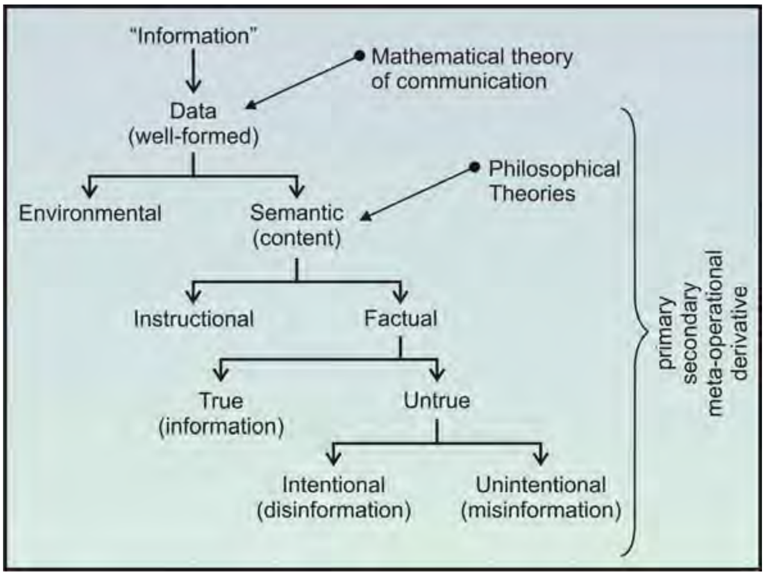


Figure 3-1 An information map (Stanford Encyclopaedia of Philosophy, 2010).

Information was formally defined in 1948 by Shannon as a tripartite between (Chiu et al. 2001; Stanford Encyclopaedia of Philosophy, 2010):

1. Quantification of information in technical problems.
2. Semantic problems that deals with the meaning of truth.
3. Influential problems where the effectiveness and impact of information has an effect on human behaviour.

The *General Definition of Information* (GDI) states that information consists of *data + meaning* (Furner, 2008; Sommaruga, 2009). Based on this, data is defined as the substance of information. According to GDI, information cannot be dataless, but it can consist of a single datum. The diaphonic definition of data (DDD) states that a datum is defined as a supposed fact regarding some difference or in uniformity within some context (Stanford Encyclopaedia of Philosophy, 2010).

The interest is not in a sequence of zeros and ones (0/1) like in the communication and computer programming industry[25]. The interest of information in this definition is based on its influence it has on the decision-making process from a philosophical and practical perspective (Furner, 2008). It is especially important to determine how information is processed in technical problems and how it influences human behaviour upon an understanding of the truthfulness of the situation to be analysed.

The effect that data and information have on the environmental decision-making process is considered in the following sections, which is applied on practical field problems in water management problems. The purpose is to determine the effect that information has on the decision-making process, with a specific emphasis on sufficiency of data and the characterisation of the decision-making process.

3.3 Data and information in the decision-making process

3.3.1 General

Experience in practical problems is that the natural tendency of scientists is to obtain as much data as possible and on as small a scale as possible in the pursuit of knowing more about the problem at hand. The ultimate purpose of collecting data should be to obtain information on which to base management decisions. Water science and management is based on quantitative methods to delineate, evaluate and understand water resources. The main purpose is to determine water quantities and qualities to be able to do prospective evaluations and to plan ahead for the purposes of resource management.

3.3.2 Data and information

When data is analyzed, it becomes information which upon interpretation increases the level of knowledge and understanding that is used as the basis for management decisions (Figure 3-2).

[25] This is known as information theory which focuses more on communication and signalling (Stanford Encyclopaedia of Philosophy, 2005).

The purpose of data collection for a groundwater assessment such as the reserve[26] must be to assist in environmental management, which needs to strike a balance between development and protection (Section 2.5). This process requires that the resource must be quantified. If e.g. excess water is available in an aquifer following the allocation to the reserve components, there is e.g. no requirement to protect the resource other than stating what volume of groundwater can still be used or allocated for development and vice versa.

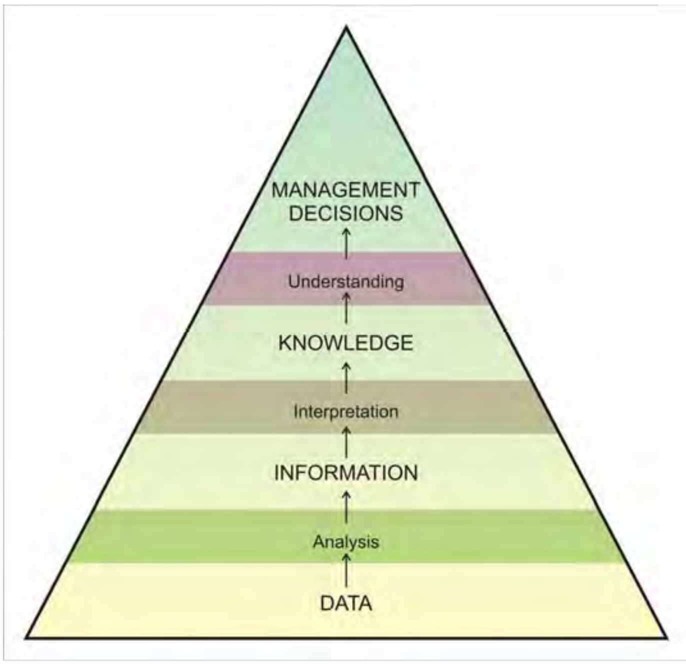


Figure 3-2 The role of data in the decision-making process (own construction).

3.3.3 Data and environmental resource quantification: A Groundwater perspective

When quantification of groundwater[27] resources is considered, issues with regard to the sufficiency of data are always an important and controversial aspect. There is a reductionist viewpoint that since the study area is of a regional nature and groundwater (point) data is scarce

[26] The water reserve is legislated in the National Water Act (NWA, Act 36 of 1998) as the basic human need and environmental water requirements that have preference and which must be allocated out before other uses are considered (Information box 4-A).
[27] Groundwater is used as an example in this study. The trends characterised here should be applicable to other environmental specialist components.

and or unreliable, an attempt to quantify the resource would be in vain. This viewpoint also implies that more data on a smaller scale is better as it is expected to reduce uncertainty. In most groundwater problems, data is sparse, which leaves the analyst with a *no solution* as it advocates that the resource should not or cannot be quantified unless *sufficient* data is available. Where data do exist, it is often treated as *exact*. A lot of emphasis is put on the *actual field data* as if it is deterministic in nature, without due consideration of the spatial and temporal variability, uncertainty and data validity (Sections 2.9.2 and 2.9.4). A typical viewpoint is that boreholes are the most important groundwater sink or role player in resource quantification. This leads to an over-analysis of borehole data and a negligence of other potential sinks, such as alien vegetation or wetlands that could be of more importance (AGES, 2010b). It also leaves the regulator in an impossible scenario as the time that it would take to collect data is often too long, which leads to the problem of doing research or collecting data *while the house is burning*.

The concept of using a decision making framework approach to make management decisions, as opposed to purely relying on technical data and assessment was proposed by Janse Van Rensburg (1992). It was re-iterated by Van Blerk (2000), who showed that a pure technical approach based on the collection and analysis of groundwater and other data at the National Radioactive Waste repository at Vaalputs, lead to more questions than answers when done outside a decision-making framework. Decision frameworks that makes use of scientific processes such as quantitative models provide valuable information.

To illustrate the problems that are encountered with data, (such as field and borehole data) and information, consider the concept of *perfect information*, in the same way that an ideal gas (that does not exist) is used in physics to demonstrate the physical properties of gases (Beuche, 1986). One would be able to make the perfect decision from *perfect information* (Dakins et al. 1993; Wisniewski, 2002). In the absence of perfect information, analysts have a problem, which is to identify when data is *sufficient* in providing information for decision-making.

3.3.4 Characterization of data and information: Applied in a groundwater context

As discussed in the previous section, when environmental data is gathered, there is a viewpoint that *more is better*. The aim of this section is to determine whether this is the case, based on data gathered from field sites. Groundwater case studies from three sites namely; The Middelburg Site (AGES, 2010c), Kalahari (AGES, 2010d) and Sand River (AGES, 2010e) Sites were used where boreholes were drilled to characterise aquifer systems (Figure 3-3).

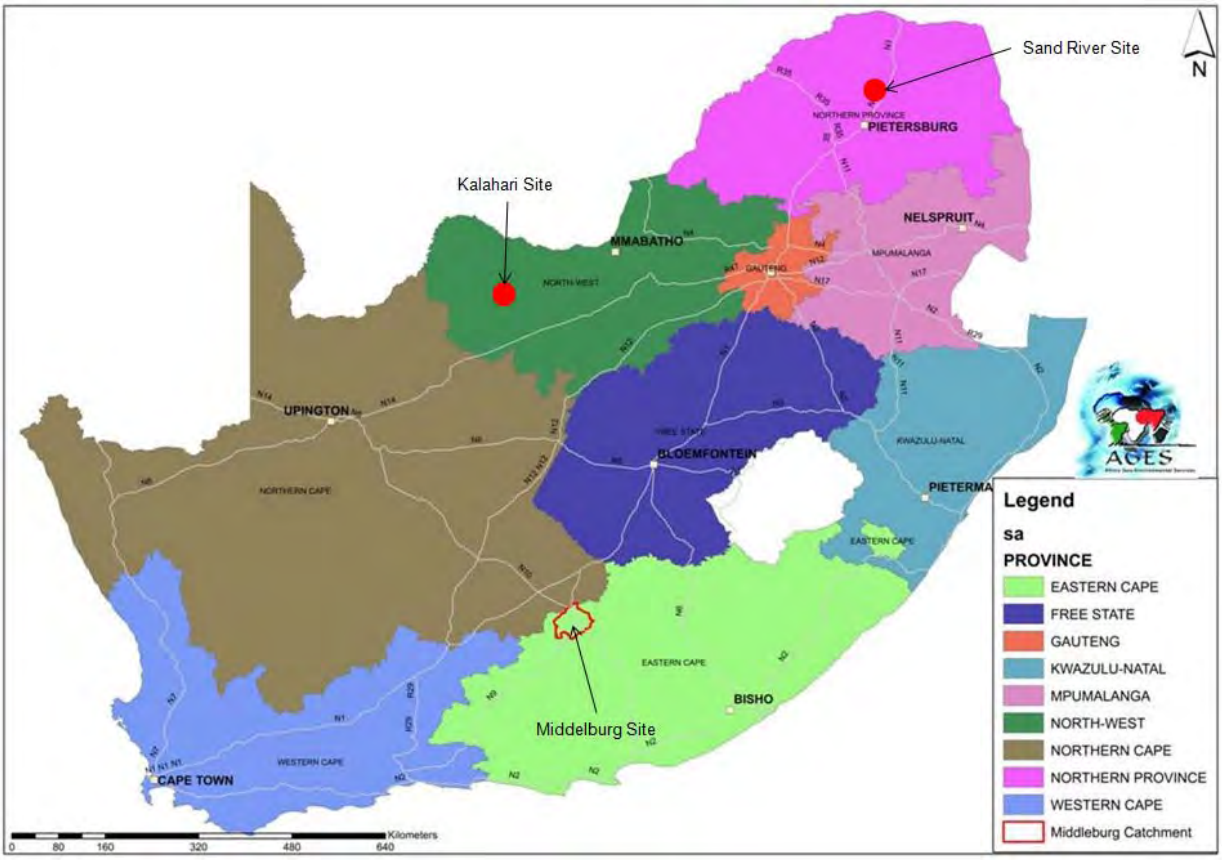


Figure 3-3 Location of the 3 field sites.

When a new aquifer system is developed, where say no prior information is available, new boreholes are drilled. Let's consider that the depth to the groundwater level is the variable that is required to be analyzed (it could be any other variable such as transmissivity etc). The first borehole is considered to provide the most information as there is no prior information available. The second and third boreholes would continue to do the same and so on. It must be established whether the data that is gathered leads to *sufficient information* at some point for the purposes of decision-making (Figure 3-4). Information can be defined here as an *accumulation and arrangement of data* (i.e. cumulative data). Data is seen as analogue to pixels and information is the *picture* that is formed from the accumulation and arrangement of the *data pixels*. For this purpose, the data from three field sites were considered where existing boreholes were surveyed and new boreholes drilled to determine the depth to groundwater level.

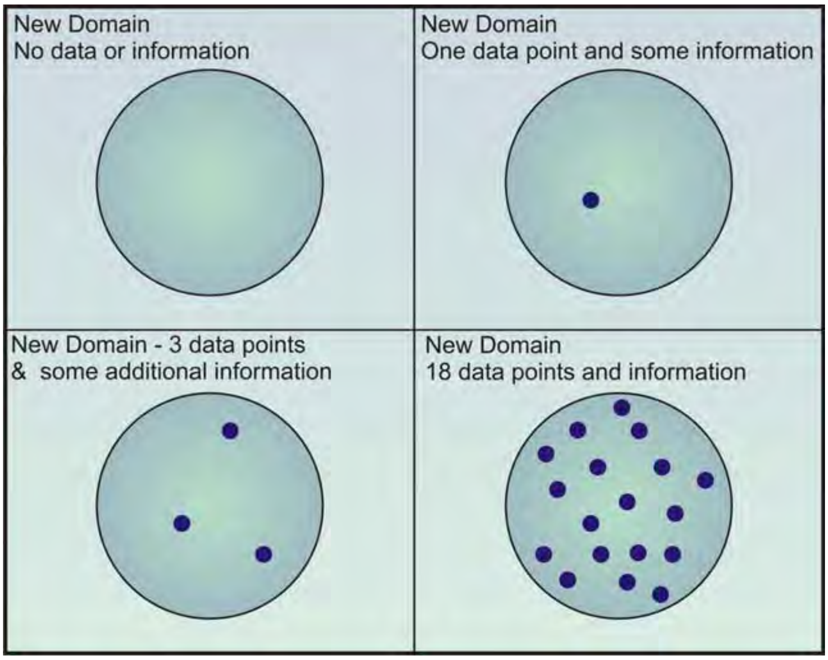


Figure 3-4 Schematic representation of the data gathering process in a new, unknown domain (own construction).

It is considered that information regarding the depth to groundwater is determined by the average and the standard deviation. If the additional data points do not contribute significantly to the change in the average or the change in the standard deviation, then the data points for the area is considered as *sufficient. Perfect data* is then defined as data with a standard deviation of zero,

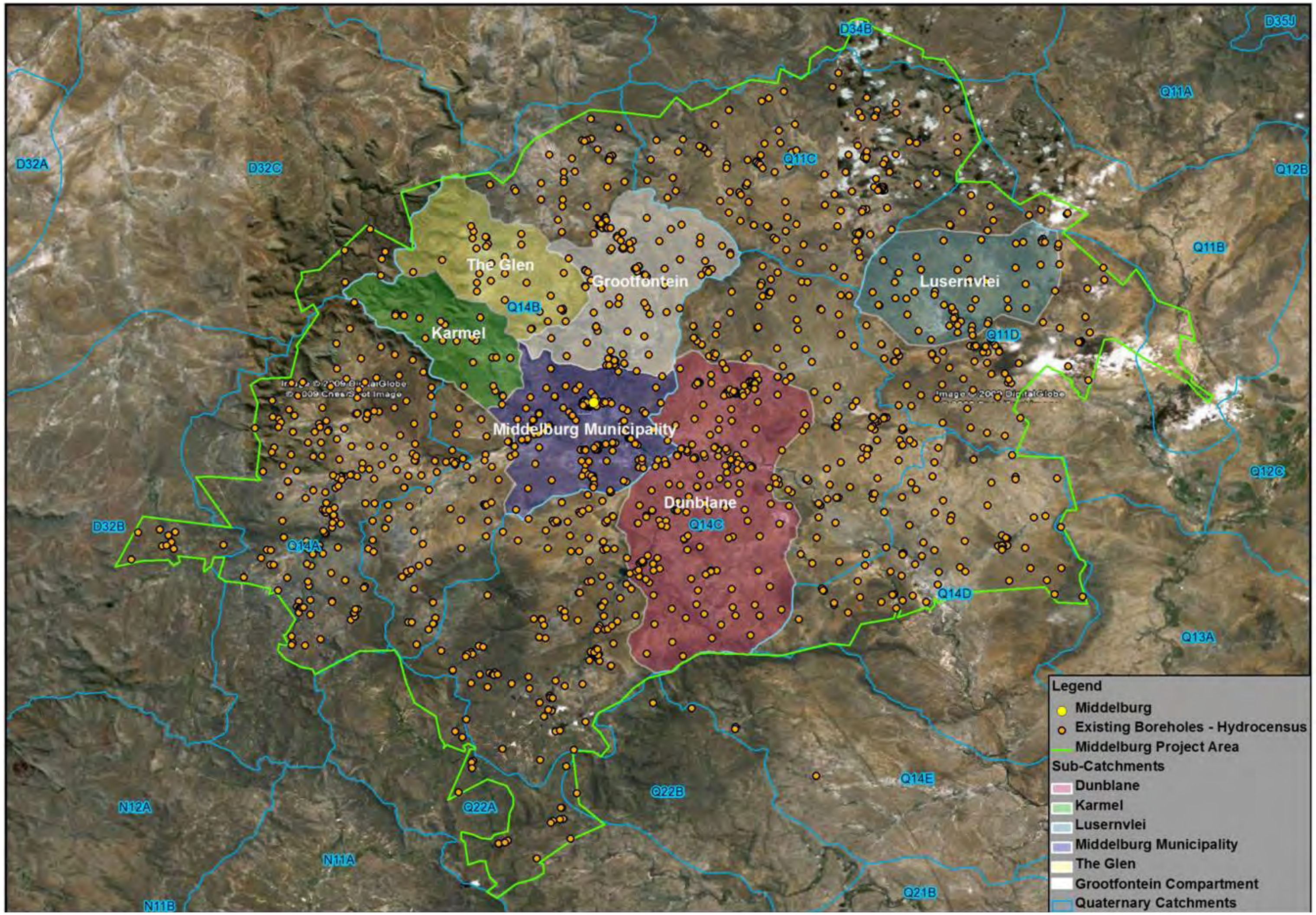


Figure 3-5 Middelburg – catchment map showing the borehole distribution (AGES, 2009).

which like in the case of perfect information does not exist in nature. If the standard deviation would be zero, the analyst require only one data point to obtain perfect information and would have no uncertainty in the data, as all the points would have the same value. Information is represented by the change in e.g. the average depth to groundwater for each additional accumulative data point that is gathered. If this value does not change significantly, then no significant information is gained, despite the fact that more data is gathered.

3.3.5 Data and information analysis: The Middelburg Site

To investigate the role that data plays in painting the information picture, the Middelburg Site was evaluated. The Middelburg study area covers 3 Quaternary Sub-Catchments (Q14A, Q14B and Q14C) in the Eastern Cape, covering a surface area of 2052 km^2 (Figure 3-3, Figure 3-5). The purpose of the project was to supply groundwater to the town of Middelburg in the Eastern Cape Province (AGES, 2010c; Annexure A).

The first borehole surveyed at the Middelburg Site (Figure 3-3) indicated a water level depth of 12.88 m. The second borehole had a water level of 12.26 m with an average between the two points of 12.57 m. The third borehole had a water level of 12.42 m with a cumulative average of the three points of 12.52 m, the fourth borehole had a water level of 6.47 m that changed the cumulative average between the first 4 points to 11.01 m, and so on. There are 715 data points with a borehole density of, 2.86 km^2/borehole. A graph of the 715 data points (Figure 3-5) shows that the cumulative change in the average and the standard deviation converges with the number of data points (Figure 3-6). As it is accepted that perfect information is not attainable, the analyst must decide on a maximum error or convergence value that would be considered as *acceptable* or *sufficient*. In the case of the Middelburg Site, the average depth to groundwater converges towards 10.95 m and the standard deviation towards 6.9 m after 715 data points. The coefficient of variation (COV)[28] is high at 63%.

[28] The COV is a measurement of the variability of the data and represents the standard deviation as a fraction of the average (http://www.businessdictionary.com/definition/coefficient-of-variation.html).

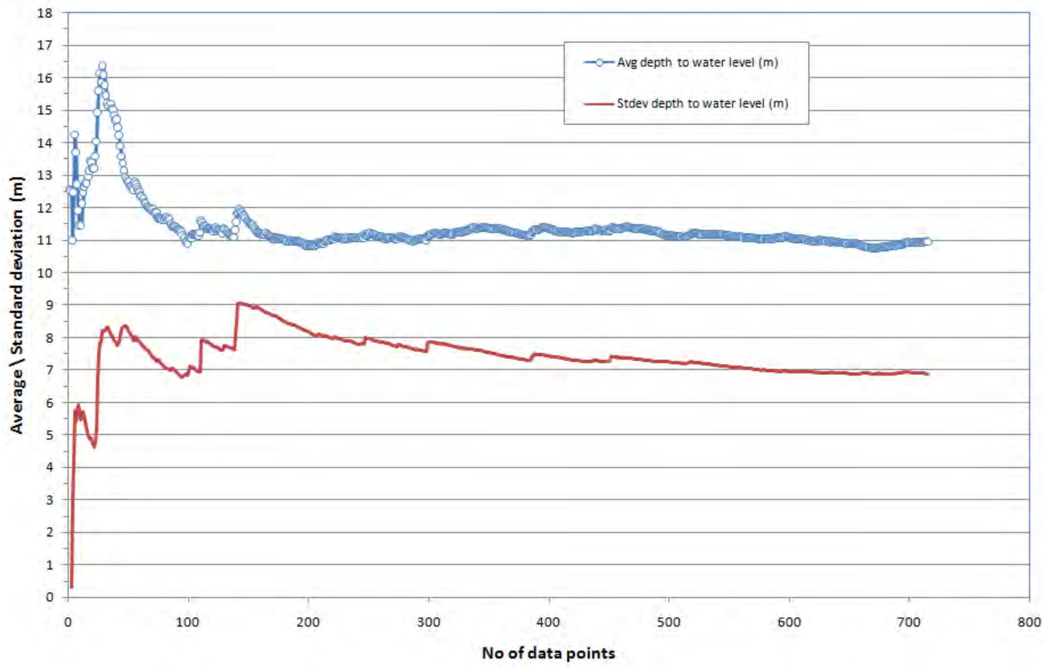


Figure 3-6 Middelburg Site depth to water level: Graph showing the cumulative average and cumulative standard deviation with increasing data points.

To determine the change in the variables with the number of data points, the first derivative was determined for both the change in the cumulative average and the change in the cumulative standard deviation with increasing number of data points. The trend shows an exponential decrease in change with an increase in the number of data points (Figure 3-7).

Except for some outliers, after 6 data points, the change in the average water level value is effectively less than 1 m (representing an error of <9 % of the actual average of 10.95 m). The change in the standard deviation is smaller than 1 m after 25 data points and smaller than 0.5 m after 30 data points (Figure 3-7). An error below 0.1 m would require more than 500 data points (Figure 3-8).

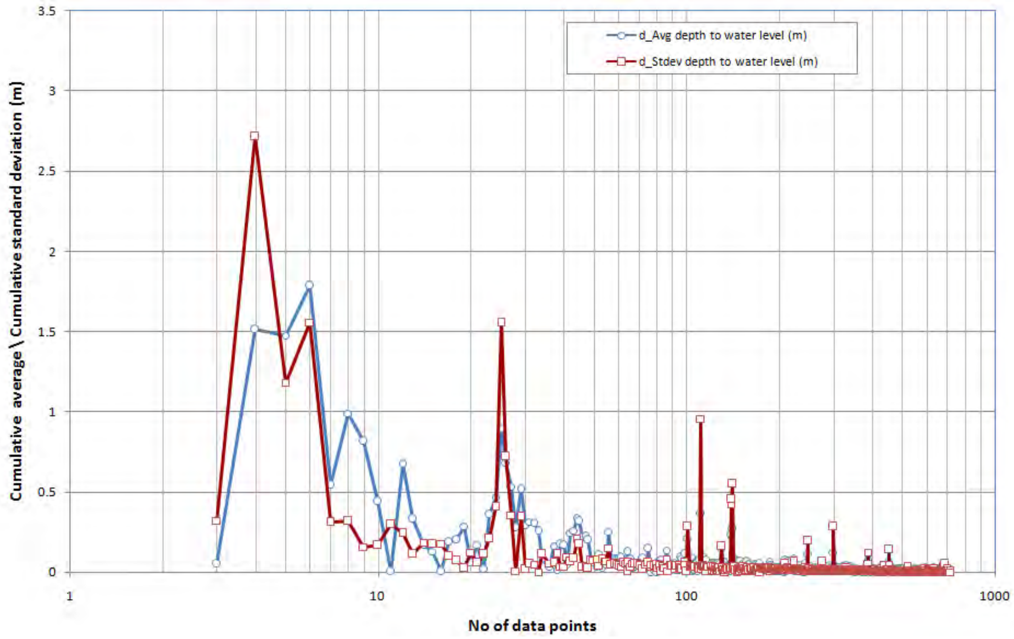


Figure 3-7 Middelburg Site depth to water level: Semi-log graph showing the change in the cumulative average and the change in the cumulative standard deviation with increasing data points.

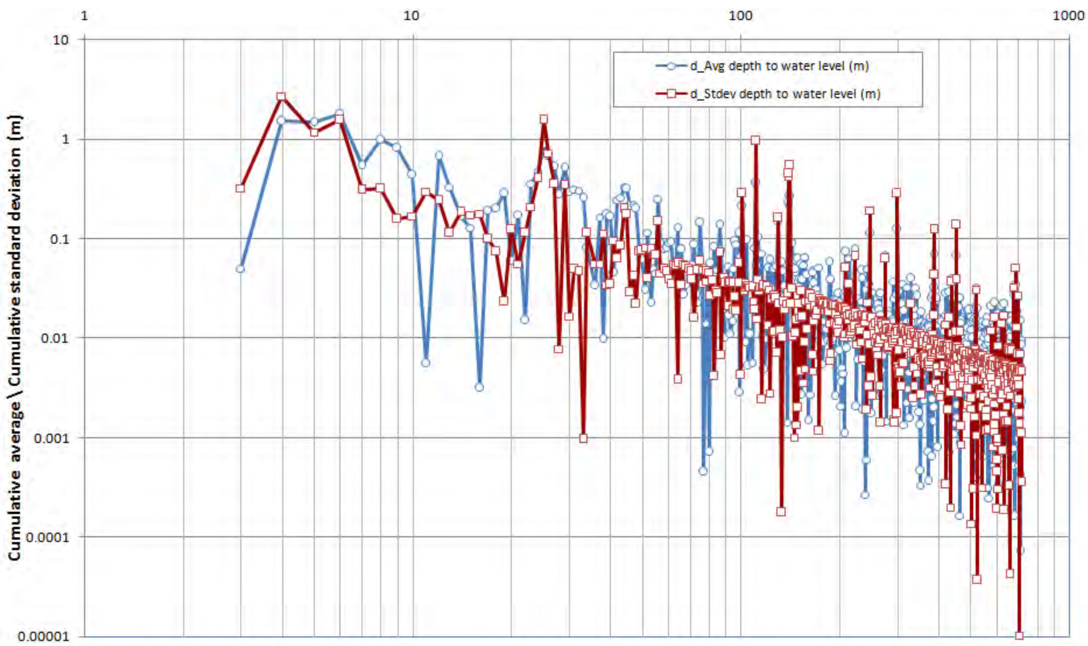


Figure 3-8 Middelburg Site depth to water level: Log-log graph showing the change in the cumulative average and the change in the cumulative standard deviation with increasing data points.

The analyst may also use the error determined as a percentage of the cumulative value, for a decision point as to when *sufficient* data was gathered (Figure 3-9). Consideration of the percentage error relative to the actual average would indicate when the error is e.g. smaller than 10% or 5%. An error of less than 5% is reached after 25 data points are gathered. Although it approaches zero, it will never reach it (Figure 3-8, Figure 3-9). Exponentially less information is provided with an increase in the number of data points (Figure 3-10).

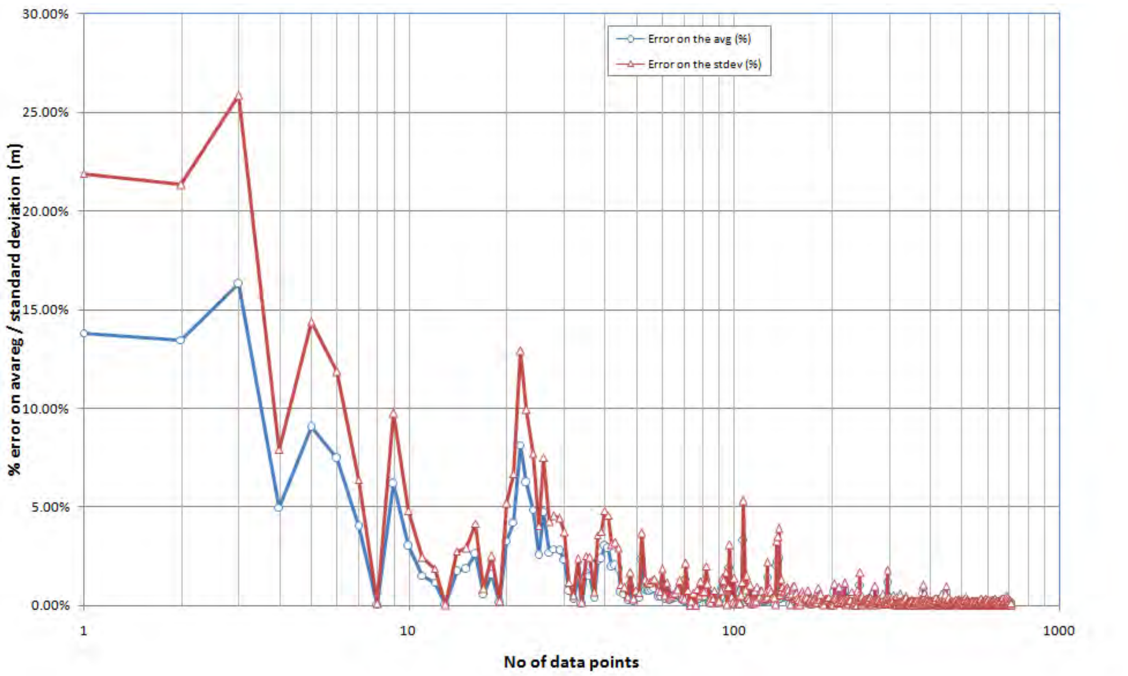


Figure 3-9 Middelburg Site depth to water level: Semi-log graph showing the % error relative to the actual value.

The information trend is similar to the *law of diminishing returns* used in economics (Mohr and Fourie, 2004). It forms a very good correlation ($R^2 = +0.95$) with a logarithmic trend on a linear plot and straight line on a semi-log plot that is defined by (Figure 3-11):

$$y = a\ln(x) - b \qquad (3\text{-}1)$$

where, a is the number of data points and b represents the *information index*. The *information index* is defined as the value towards which the cumulative change in the variable converges on the y-axis (Figure 3-10). This value represents the cumulative change in the average and the cumulative change in the standard deviation of the water level and hence the information that is provided by consecutive data points (Figure 3-10, Figure 3-11). Data points that do not significantly change the average value or the standard deviation of the variables, *do not increase the level of information and knowledge about the variable in consideration* (Section 3.3.2)

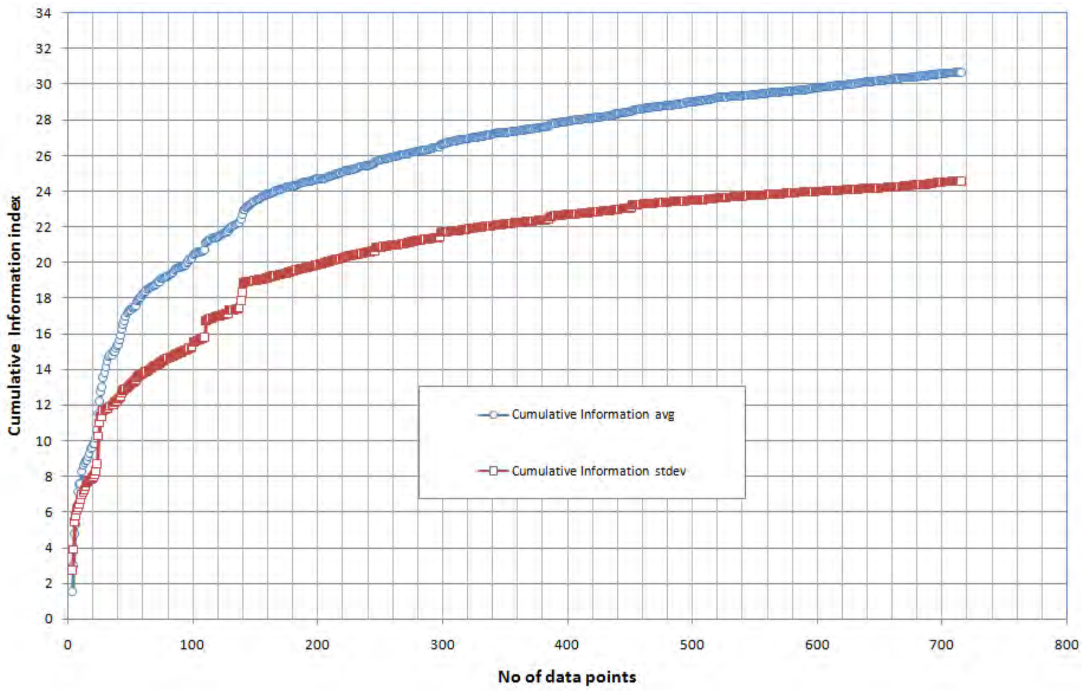


Figure 3-10 Middelburg Site depth to water level: Cumulative information graph.

Perfect information[29] can now be defined as data with an error of zero and hence zero uncertainty, which like in the case of perfect data does not exist. It does however provide a limit that can be used as a reference. Information was previously defined as an accumulation of data

29 Perfect information is defined as an illustrative reference of when information is 100% accurate. It would be represented by a horizontal line on the linear trend plot (Figure 3-10).

(Section 3.3.4). It can be determined mathematically as the cumulative change in the variable represented by the first derivative:

$$I_{ind} = \sum_{i=1}^{n} \frac{dv_i}{dn_i} \tag{3-2}$$

where *n* are the number of variables *v* from *i* to *n* and I_{ind} represents the information index. (Figure 3-10). Perfect information is reached when the gradient of the information curve becomes zero (i.e. a horizontal line), which will not happen in practice. The semi-log plot of the information curve shows that it assumes a straight line (Figure 3-11). The identification of a straight line on a semi-log plot of the first derivative could serve to identify when data becomes *sufficient information*.

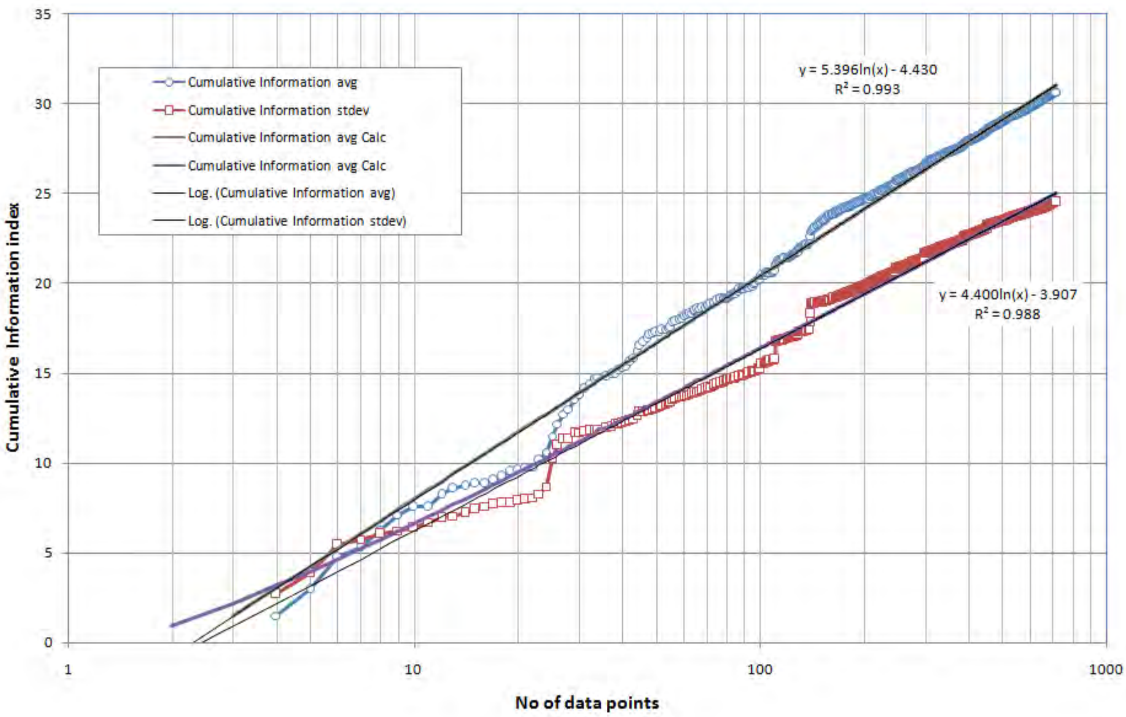


Figure 3-11 Middelburg Site depth to water level: Semi-log plot of cumulative information graph.

The cumulative line can be called the *information curve*, which converges to a value of 35 on a linear plot (Figure 3-12). As discussed earlier, the shape of the information curve indicates that less and less is gained in terms of information as the number of data points increases (Figure 3-10, Figure 3-11). Apart from the fact that the analyst may choose a *minimum* error in terms of the actual value (Figure 3-7, Figure 3-8) or as a percentage of the expected value (Figure 3-9), the question arises, when is data *sufficient*, *near-optimal* or *optimal*? (Section 2.9.7). To evaluate this, four straight gradient lines were fitted to the semi-log plot to determine the change in information with the increase in the number of data points (Figure 3-12). The aim is to determine when additional information becomes insignificant if compared to the number of data points already considered. In other words the analyst wants to determine when no more significant information or knowledge is gained about the variable under consideration.

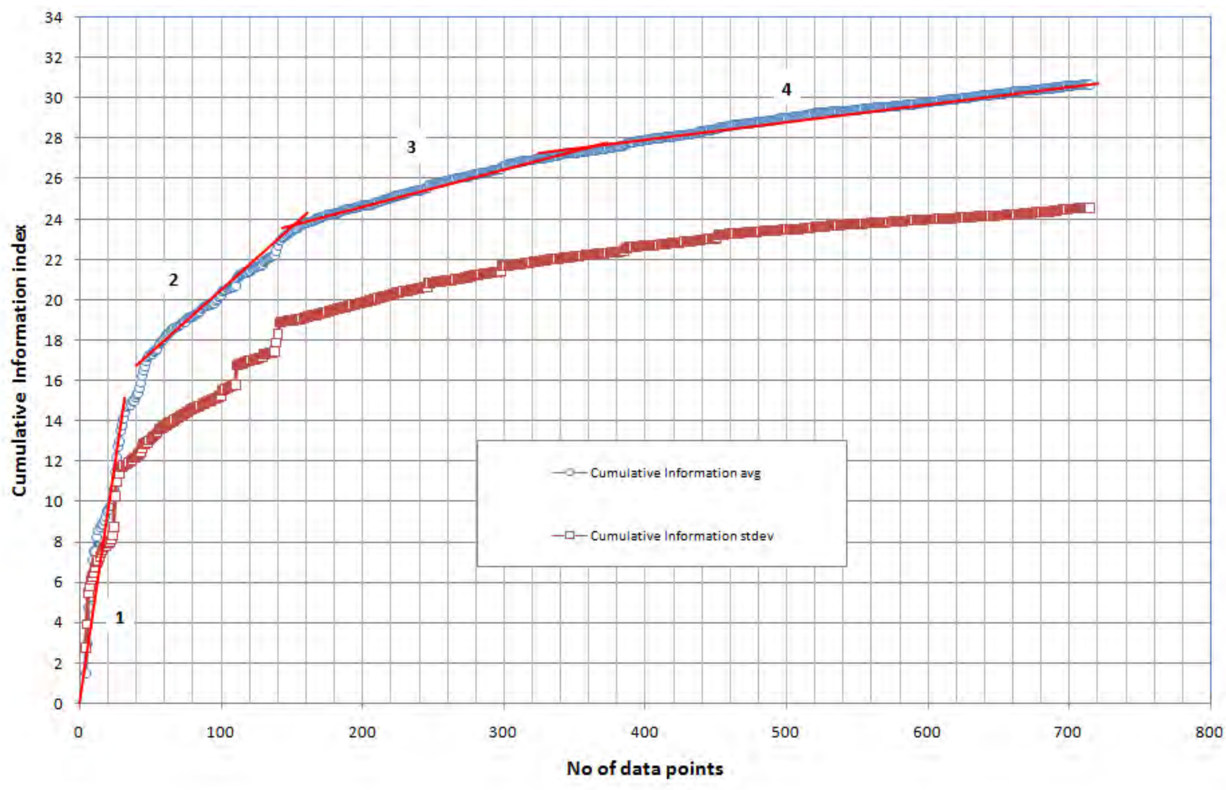


Figure 3-12 Middelburg Site depth to water level: Cumulative information graph with straight lines fitted.

Table 3-1 Middelburg Site data and information index on graph sections (Figure 3-12).

Section	Information gained	From to	No of data points	Information Index
1	15	0 to 30	30	0.50
2	7	40 to 160	120	0.06
3	3	170 to 360	190	0.02
4	3.5	380 to 715	335	0.01

The gradient lines indicated that (Figure 3-12, Table 3-1):

- The first line from 0 to 30 provided an information index[30] of 0.5.
- The second from point 40 to 160 with 0.06.
- Third line from 170 to 360 at 0.02.
- Fourth line from 380 to 715 with 0.01.

The first line has an index of 0.5 and the second line provides a difference in the known information of 0.06, which is almost 10 times less and requires 4 times the number of data points. The last gradient line provides an index of only 0.01 with 335 points which equates to 50 times less information with 11 times more data points.

If the analyst considered a 10% (or a ±1 m) error as acceptable, then only the first 6 to 10 data points would provide *sufficient* data. The first line from zero to data point 30, is considered as *optimal* data and could be seen as analogue to the rational model for decision-making (Section 2.6.2). Data points in excess of 50 points would only add value to reduce the error margin below 0.25 m or 3 % (Figure 3-7, Figure 3-8, Figure 3-9).

3.3.6 Kalahari and Sandriver sites

Similar assessments were done on 2 alternative field sites (Figure 3-3) with different

[30] The information index is defined as the change in the variable with an increasing number of data points, which represents how much we know more against the background of what we already know with an increasing number of data points.

groundwater conditions to determine if the same behaviour in the information with increasing number of data points is obtained.

The second site is in the Kalahari, north of Stella in the North-West Province. This site is located in an arid area with a very flat topography. The geology consists of surficial Kalahari Sand deposits that vary in thickness from 5 m to 20 m. The Kalahari Sand is underlain by Greenstone Formations from the Kraaipan Group. The site contains 203 boreholes with water level data in an area covering 835 km^2. The borehole density is 4.1 km^2/borehole and there is only minor abstraction from groundwater taking place for the purposes of domestic use and livestock watering.

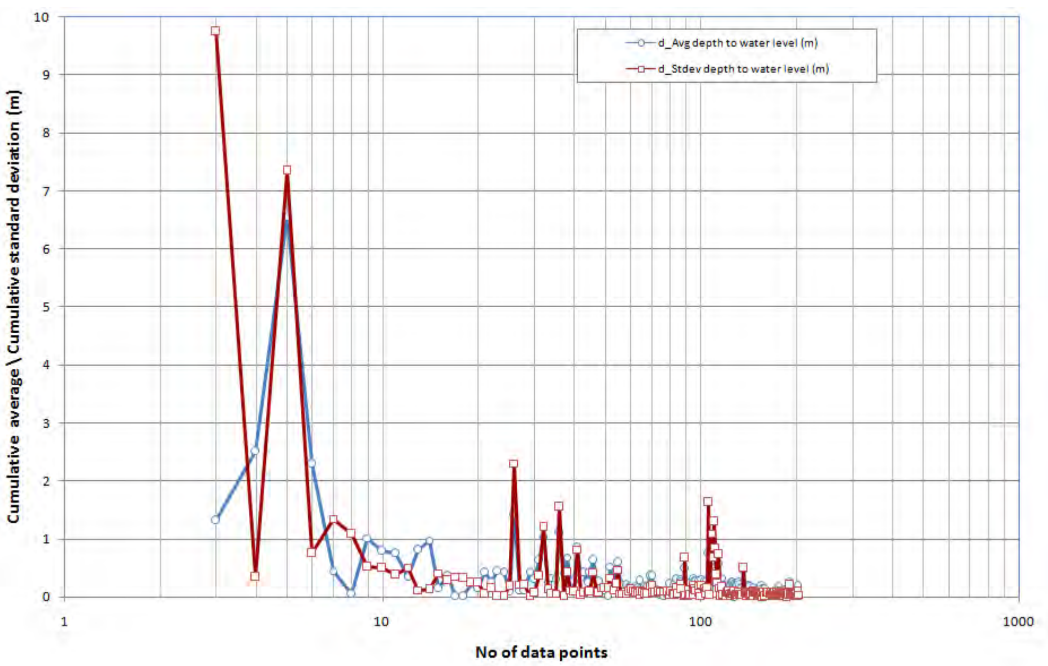


Figure 3-13 Kalahari Site depth to water level: Semi-log graph showing the error relative to the number of data points.

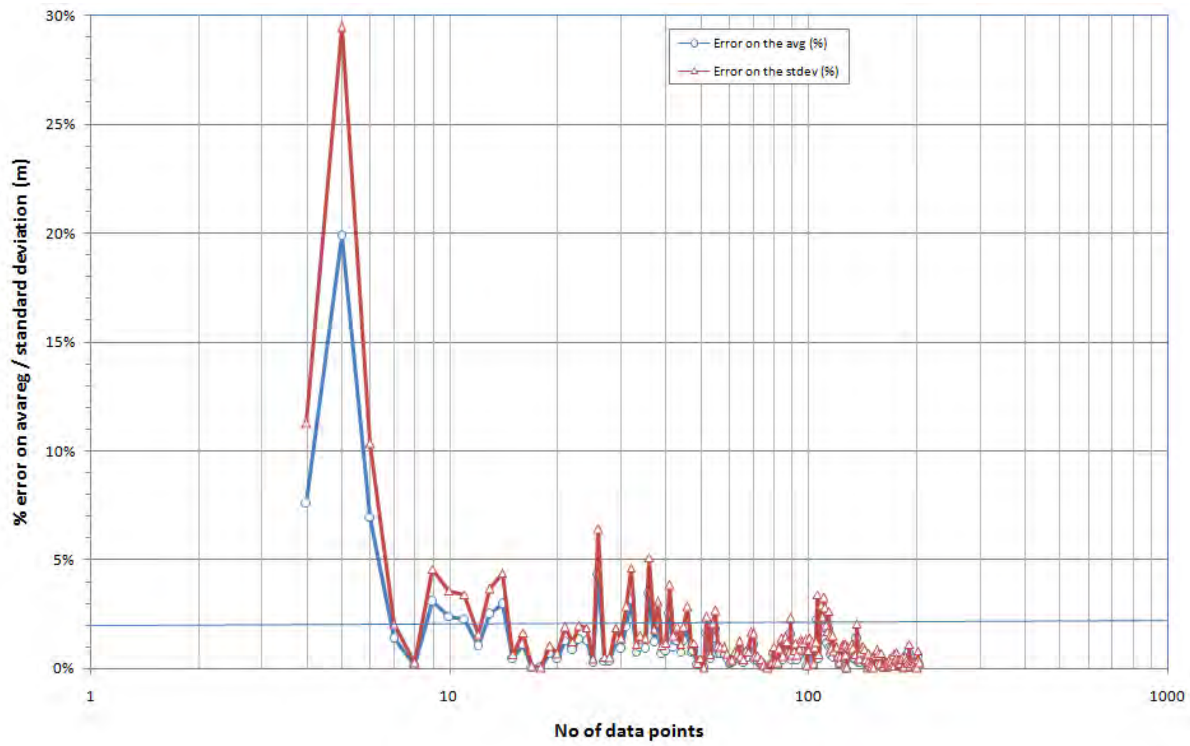


Figure 3-14 Kalahari Site depth to water level: Semi-log graph showing the % error relative to the actual value.

The analysis of the average and standard deviation on the depth to groundwater level was done as in the previous Section with the Middelburg data (Figure 3-13 to Figure 3-20). The analysis indicates that similar to the Middelburg data that for the Kalahari Site, the % error in the average water level decreases to below 5% after 6 data points and to below 2% after 45 data points (Figure 3-13, Figure 3-14). The average depth to water level from the 203 data points is 32.94 m with a standard deviation of 22.20 m. The cumulative information with number of data points also follows a logarithmic trend (Figure 3-15) or a straight line on a semi-log plot (Figure 3-16). The COV is high at 67%, which indicates a high variability in the data.

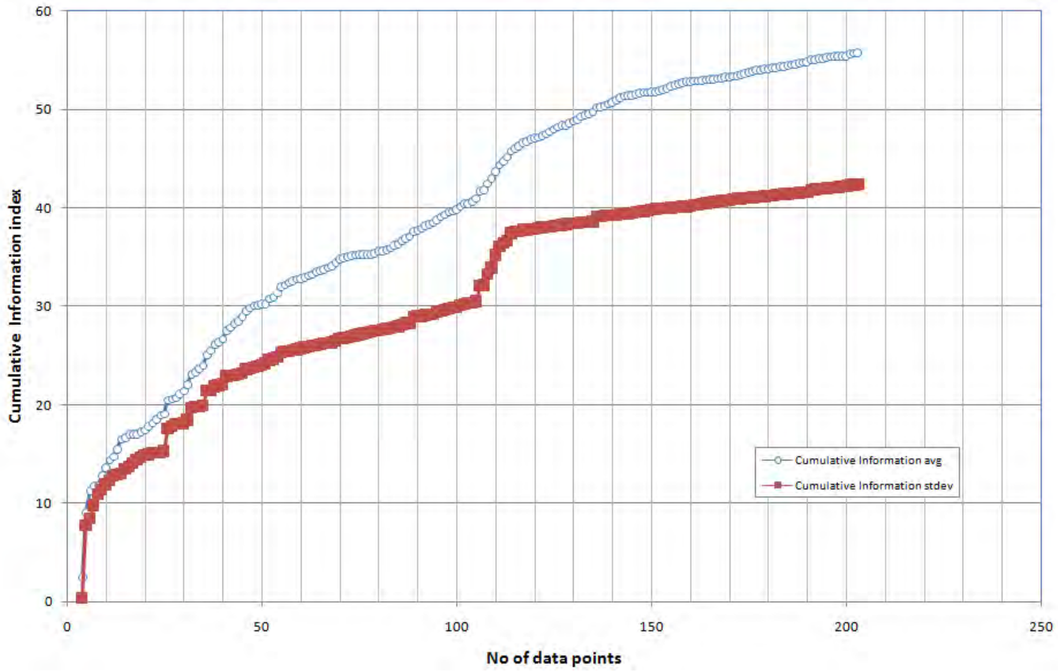


Figure 3-15 Kalahari Site depth to water level: Cumulative information graph.

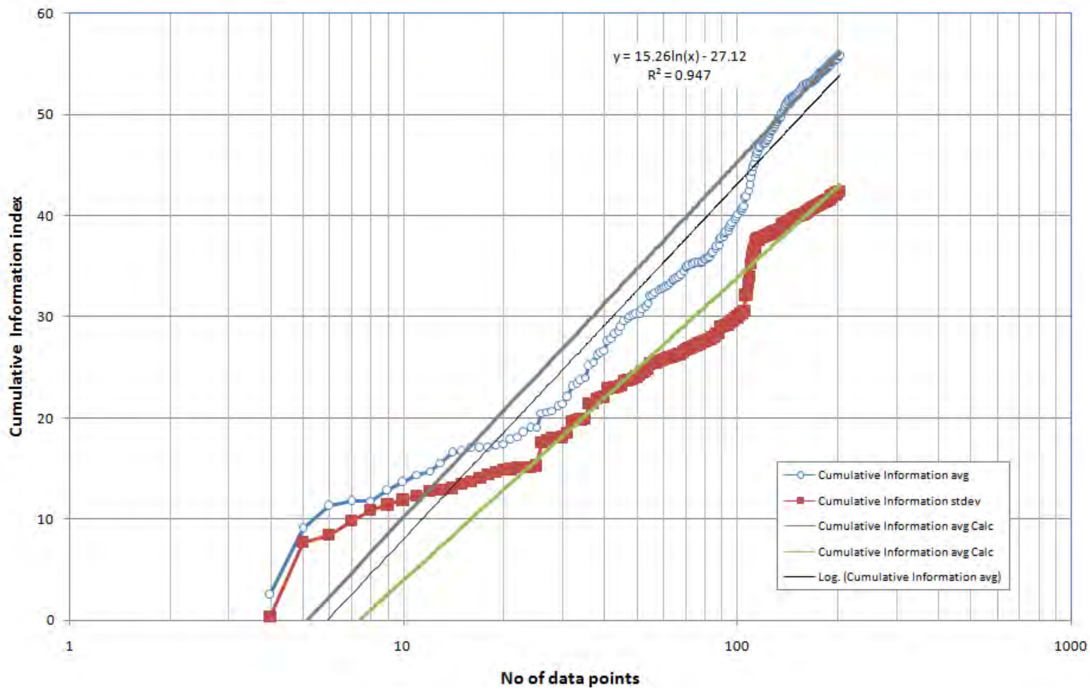


Figure 3-16 Kalahari Site depth to water level: Semi-log plot of cumulative information graph.

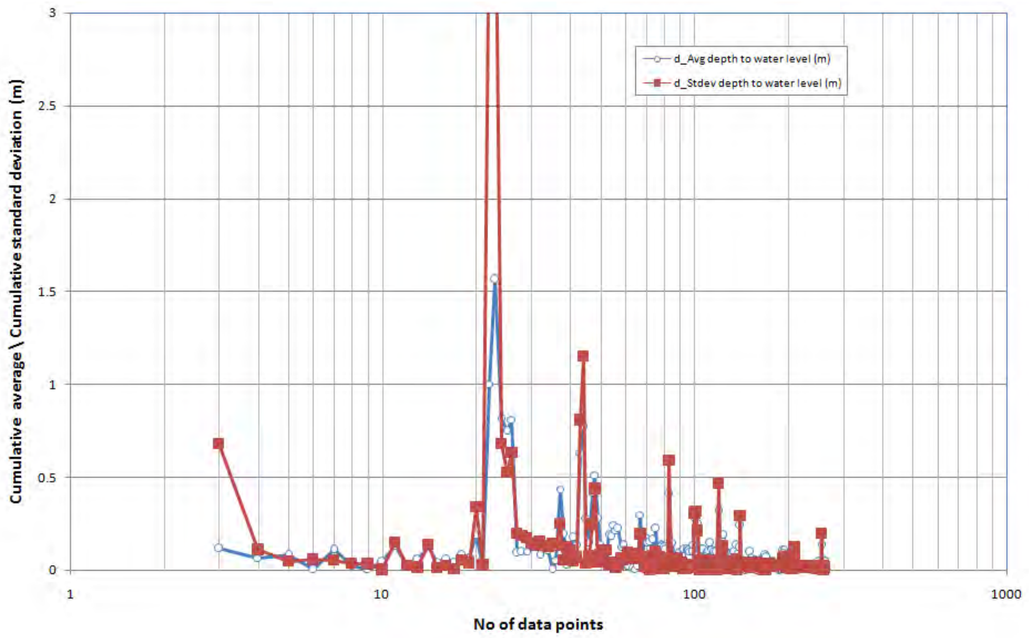


Figure 3-17 Sand River Aquifer Site depth to water level: Semi-log graph showing the error relative to the no of data points.

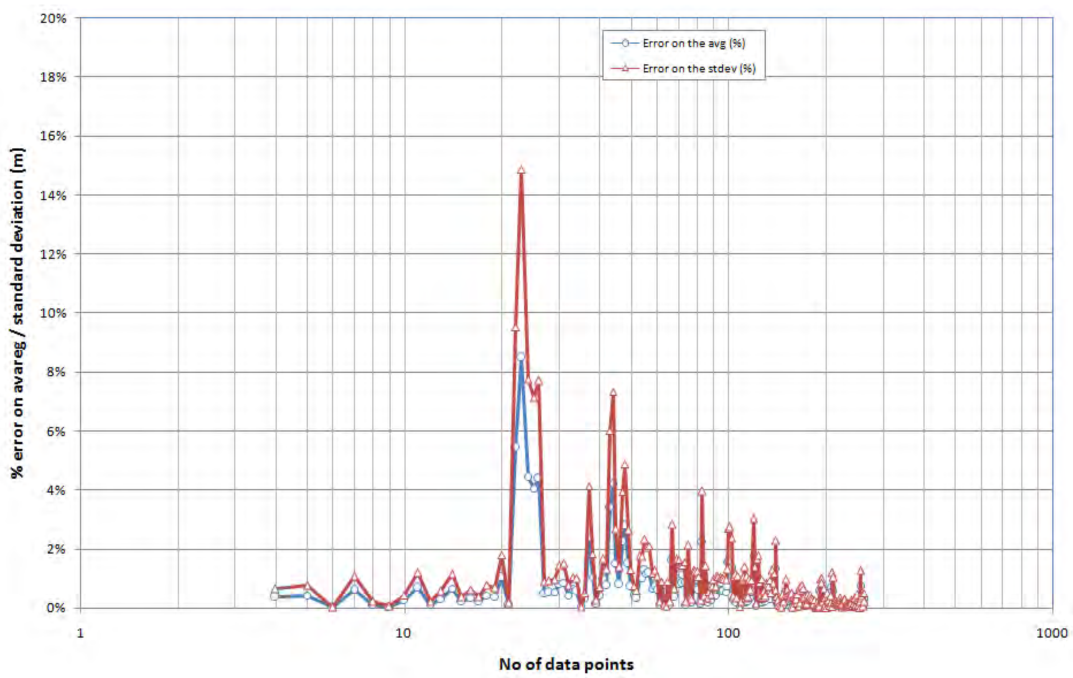


Figure 3-18 Sand River Aquifer Site depth to water level: Semi-log graph showing the % error relative to the actual value.

The third site is known as the Sand River Aquifer Site north of Polokwane in the Limpopo Province (Figure 3-3). The site covers an area of 1220 km^2 with a total of 262 boreholes with water level data. The borehole density is 4.6 km^2/borehole. The abstraction rate in the area is high at 48 550 m^3/d (562 ℓ/s) and the water is used extensively for irrigation

The analysis of the Sand River Aquifer Site showed that the % error on the average water level decreases to below 5% after 25 data points and to below 2% after 85 data points (Figure 3-17, Figure 3-18). More data points were required to provide the same magnitude in the error than the previous two sites. This can be ascribed to the high volumes of abstraction that takes place, which changes the depth to groundwater level over short distances, which is influenced by borehole distribution and utilisation. The outlier on point 23 in Figure 3-17 and Figure 3-18, is a borehole (Z11A) that has an anomalous deep water level of 43.86 m, compared to the average depth of 18.44 m. This is due to excessive abstraction from that specific borehole. Outliers were included in the assessment to indicate that the trend is still followed, even if there are outliers.

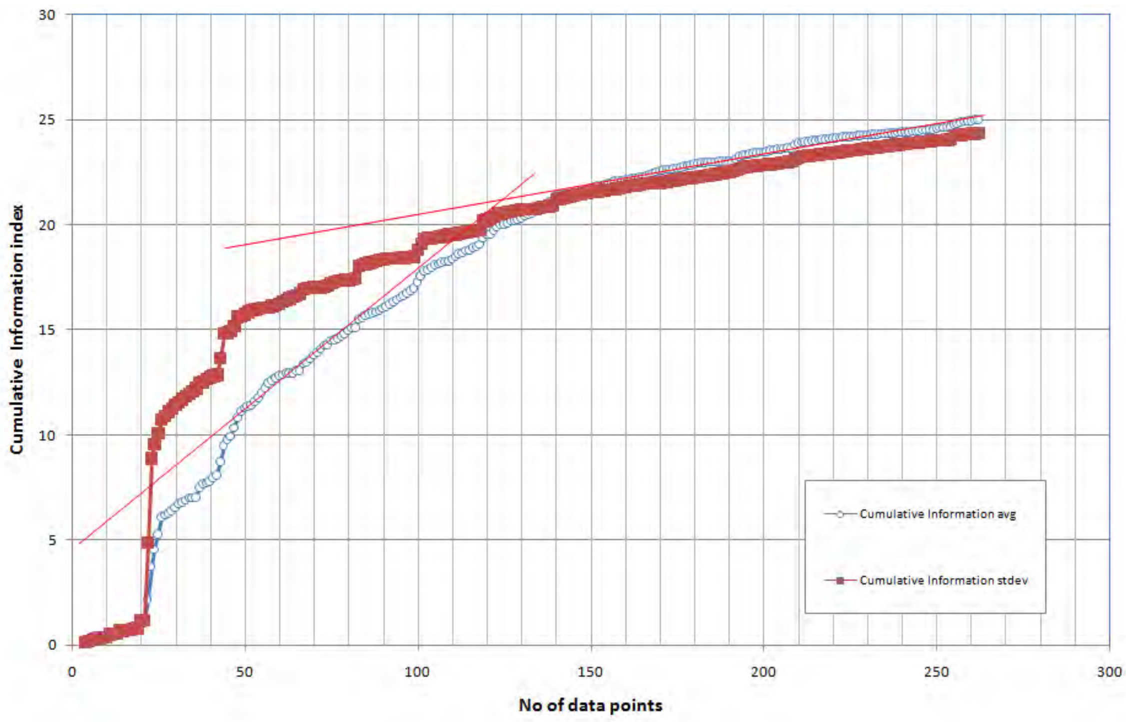


Figure 3-19 Sand River Aquifer Site depth to water level: Cumulative information graph.

The average depth to water level from the 262 data points is 18.44 m with a standard deviation of 10.52 m. The COV is high at 53%, with the cumulative information with number of data points following a logarithmic trend (Figure 3-19) or a straight line on a semi-log plot (Figure 3-20). Although the COV is high at 53%, it is still lower than the previous two sites which were at +60%.

All the analyses showed that there is a *diminishing return on information* with an increase in the number of data points (Section 0; Mohr and Fourie, 2004). *More data is therefore not necessarily better* in terms of providing more information as an optimal point is reached.

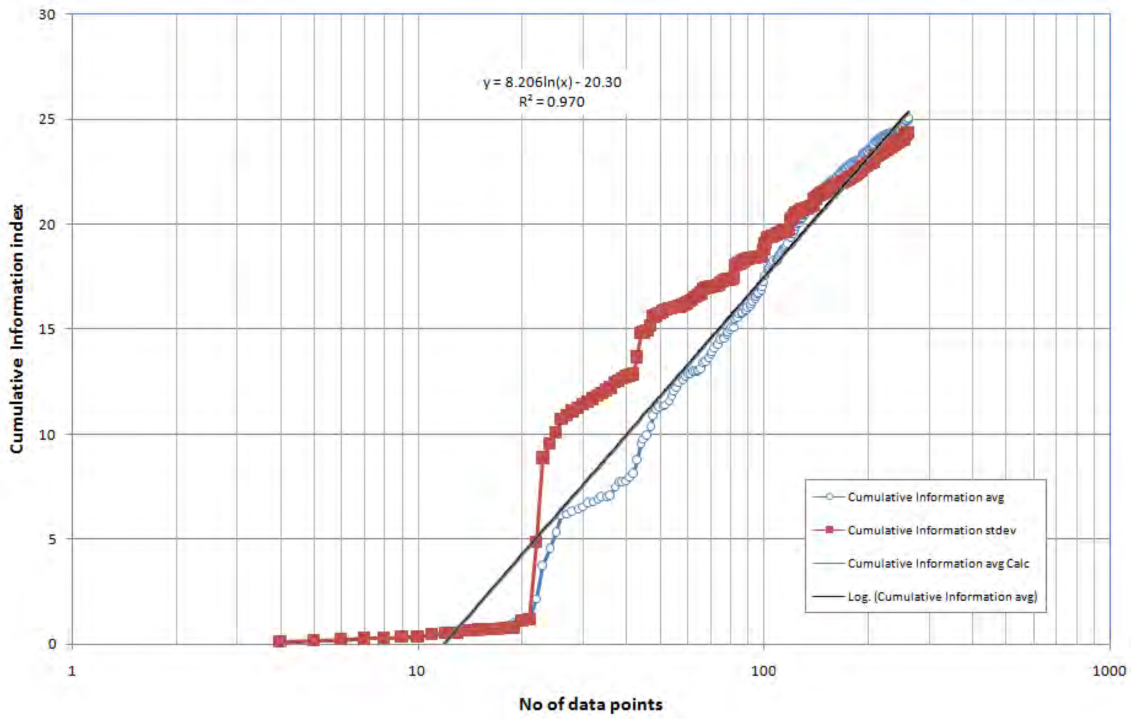


Figure 3-20 Sand River Aquifer Site depth to water level: Semi-log plot of cumulative information graph.

3.4 Geostatistical analysis: The semi-variogram

An important aspect that is associated with geological assessments is the spatial variability in parameters such as the thickness and grade of e.g. a coal orebody (Figure 3-21). For this purpose, the geostatistical semi-variogram was developed, which is given by (Clark, 1979);

$$2\gamma^{*}(h) = \frac{1}{n}\sum\left[g(x) - g(x+h)\right]^{2} \tag{3-3}$$

Where the term $2\gamma^{*}(h)$ is known as the semi-variogram, g represents the grade or orebody thickness and x is the position of one sample in a pair and $(x+h)$ the position of the other. The term γ^{*} has the same relationship to γ, than a histogram has to a probability distribution in statistics.

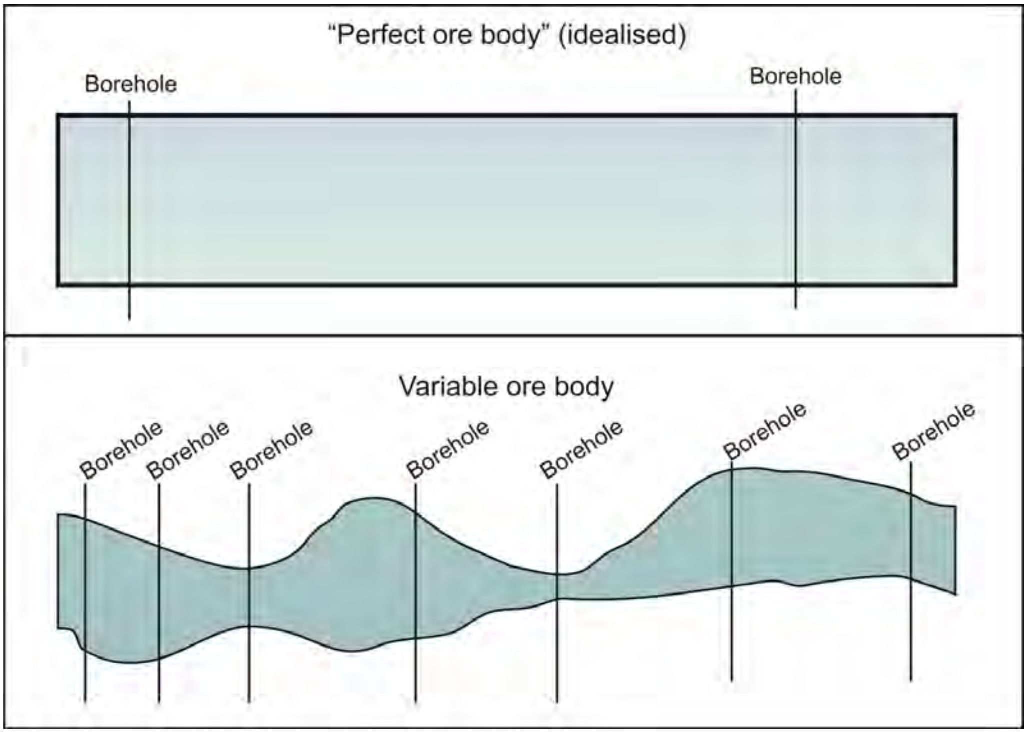


Figure 3-21 Schematic representation of an idealised perfect ore body and a variable orebody.

Although there are a number of spatial models for the semi-variogram, the spherical model illustrates the ideal behaviour of the data with distance (Figure 3-22). There are however other models that are linear or exponential (Clark, 1979). The spherical model for the geostatistical semi-variogram has a similar logarithmic trend of the site data that was plotted in the previous section (3.3), although the objectives of the processes are different. The semi-variogram shows the minimum distance between sampling points which could be boreholes or samples down a single borehole that is valid for the purposes of interpolation. This is indicated by the part of the semi-variogram where γ flattens with increasing *h*. The flattened part of the curve is known as the sill (C) and where it intersects the x-axis is known as the lag distance (a) (Figure 3-22).

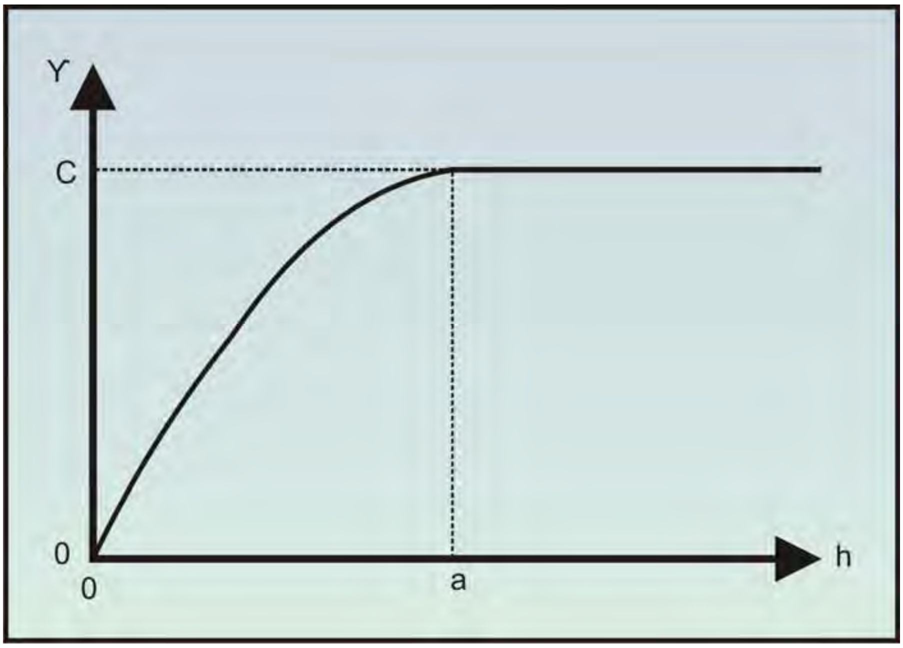


Figure 3-22 The ideal shape for a semi-variogram shown by the spherical model (Clark, 1979).

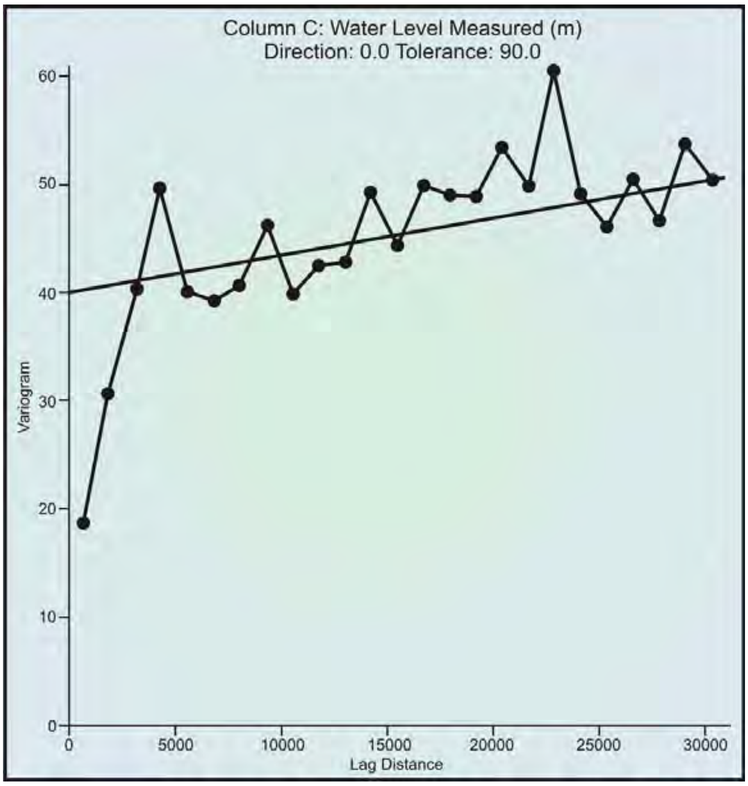


Figure 3-23 Semi-variogram for the Middelburg Site data (depth to water level).

A semi-variogram for the Middelburg depth to groundwater levels, indicates a lag distance of 5000 m. It means that based on the variability of the data, boreholes located >5 000 m from each other do not have a statistical relationship in the water level data and cannot be used for interpolation purposes. The average distance between boreholes at Middelburg is 53 m, a much smaller distance than the required maximum of 5 km, which means that the data is valid.

3.5 Data worth

The collection of data is associated with cost and time (Figure 3-25). Data therefore has value or is obtained at a cost (Freeze, et al. 1992). As data collection programmes could be never ending, data sufficiency and financial constraints will dictate when data collection would need to end. The concept of data worth relates to the value of the data that is collected in site investigations. The first borehole that is drilled or surveyed would e.g. provide the most valuable information in a catchment where there are e.g. no previous boreholes drilled (Figure 3-4). As more boreholes are drilled or surveyed, there would be a point where there are sufficient, say 30 or 50 boreholes surveyed (Section 3.3.4) so that new additions would provide less data in terms of value than the actual cost of the borehole or the survey (Freeze, et al. 1992; Dakins et al. 1995). It is therefore not sensible to aim at collection of all the data in a given catchment for a given project, but rather to *suffice* or *optimise* (Section 2.9.7) in terms of statistical representativeness and information provided.

The optimal point for data collection would be if the % error is below a preset value as determined by the analyst (e.g. <5%) or at the plateau of the logarithmic plot of information flow (Figure 3-10; Figure 3-25, Section3.3.5). To evaluate the cost of data collection for the Middelburg Site, it was determined that to collect one data point costs R200 (Table 3-2). This cost includes the time of the field surveyor that includes travelling and accommodation. The total cost for surveying 715 data points amounts to R142 800. The information gained is the difference between e.g. the average depth to water level with each additional data point gathered (Figure 3-12, Section 3.3.4).

Table 3-2 Middelburg Site data, information and cost of data (Figure 3-24).

Section	No of data points	Information gained	From to	Information Index	Cost per data point (R)	Cost per section (R)	Cost per unit information (R)
1	30	15	0 to 30	0.50	R 200	R 6 000	R 400
2	120	7	40 to 160	0.06	R 200	R 24 000	R 3 429
3	190	4	170 to 360	0.02	R 200	R 38 000	R 9 500
4	335	3	380 to 715	0.01	R 200	R 67 000	R 22 333

The cost per information unit is represented by:

$$CU = \frac{CS}{IG} \tag{3-4}$$

Where the cost per unit (*CU*) is provided by the information gained (*IG*) and the cost per section (*CS*). The analysis shows that the cost per unit gets exponentially more expensive with the number of data points. The cost per information unit for line 1 is R400 and for line 4, it is more than R22 000 (Figure 3-12). The information gained and cost can be optimised by plotting the information gained and costs against the number of data points (Figure 3-24). The *optimal*[31] number of data points to gain information based on costs is at 149 data points at a cost of R29 600 (Section 2.9.7). The gathering of more data points would result in a higher cost relative to the information gained.

Sufficient information[32] could be determined by the analyst at an acceptable error value of e.g. <5%, which is at 30 data points (Figure 3-9, Figure 3-24). The cost of sufficient information after 30 data points is R6 000, which is substantially less than the R29 600 of the cost of the optimal no of data points.

Based on the cost and information considerations, it would be sensible to scale data collection programmes to start with aiming to obtain *sufficient* and then *optimal* information. It is therefore not true that more is better during data collection programmes. From a sustainability and practical perspective, the value of the data vs the information gained should be considered first.

[31] This approach would be followed in the rational method for decision-making (Section 2.6.1)
[32] This approach would be followed in the bounded rational method for decision-making (Section 2.6.2).

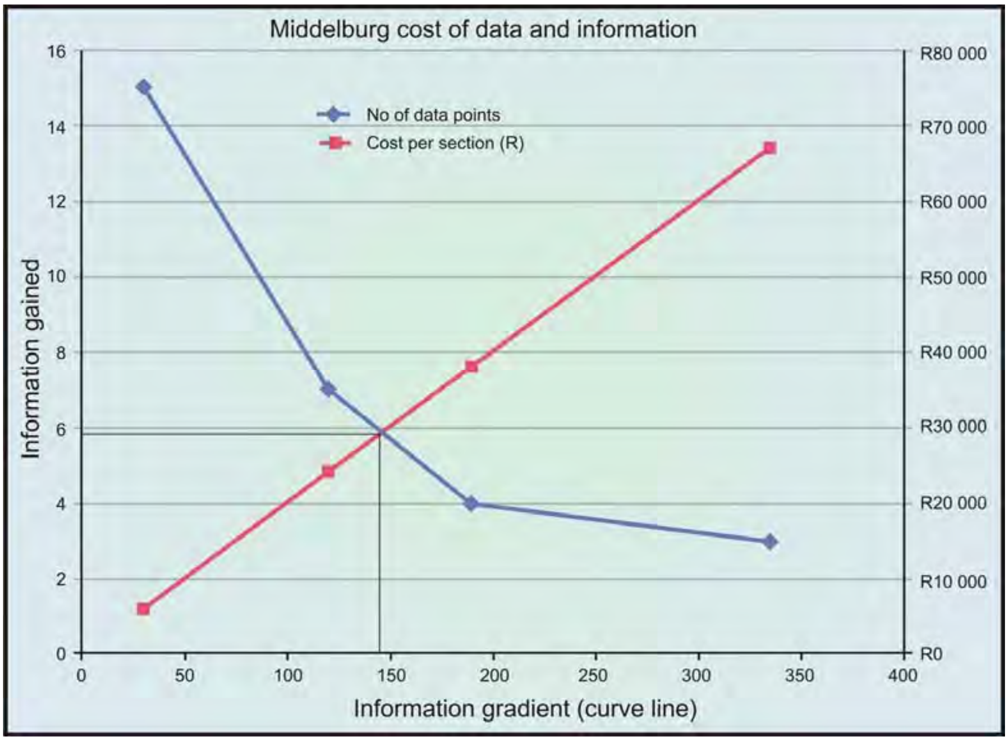


Figure 3-24 Middelburg Site: Information gained vs cost of data with optimal point.

3.6 The decision-making process

The flow of information determines the nature of the decision-making process. Too little information and the decision-maker has to make a call based on what is available (i.e. satisfice; Section 2.6.2) and too much could lead to a waste of money. In this case, it is analogue to a situation where there are not enough data pixels to produce an information picture. Too much information could lead to unnecessary expenses in terms of time and cost where there are sufficient data pixels to produce an information picture, which then becomes overpopulated when unnecessary data gathering is continued (Goldratt, 1994). The logarithmic nature of the information process flow means that there are points of *sufficient* and *optimal* information to base decisions on and that information decreases exponentially with an increase in data points. A problem that is often encountered in geohydrological and other environmental investigations is that the analyst either has too little data or aim to obtain as much data as possible in the pursuit of *perfect information*. To aim and obtain perfect information is not possible as it would amount to infinite time and cost. The data gathering viewpoint is based on the assumption that *more is better*, which leads to the proverbial *analysis paralysis* problem that is usually taken by risk-averse analysts.

The analysis of the decision-making process indicated that the analyst must understand the decision-making process and plan the data gathering exercise accordingly. If data is not gathered for the purposes of decision-making, then the data gathering exercise is futile[33]. The decision-making process indicates that it is possible to obtain e.g. 60% information with a certain amount of time (effort) and cost (equal to X), but to get from 60% to 80% certainty in terms of information, could be double in terms of time and cost (+2X) (Figure 3-25). From there it becomes exponentially more difficult to gain additional information and it would take much more to get to e.g. 80% information, while to get to 100% or *perfect information* is impossible (Figure 3-25).

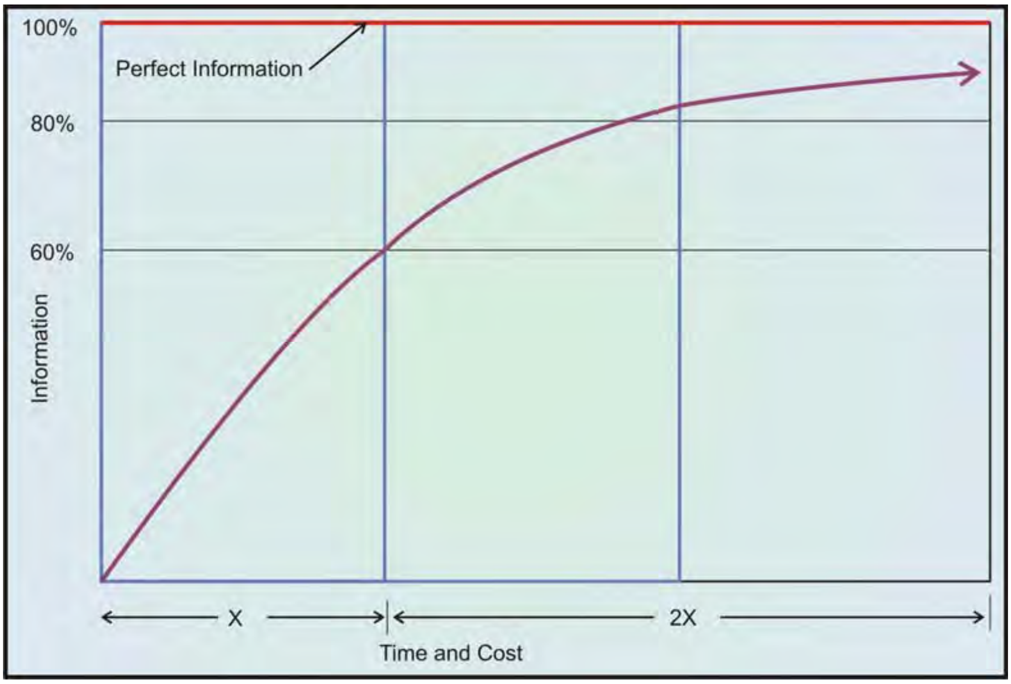


Figure 3-25 Time, cost and information flow in the decision-making process.

[33] There may be exceptions in research projects, but this study is aimed at practical cases.

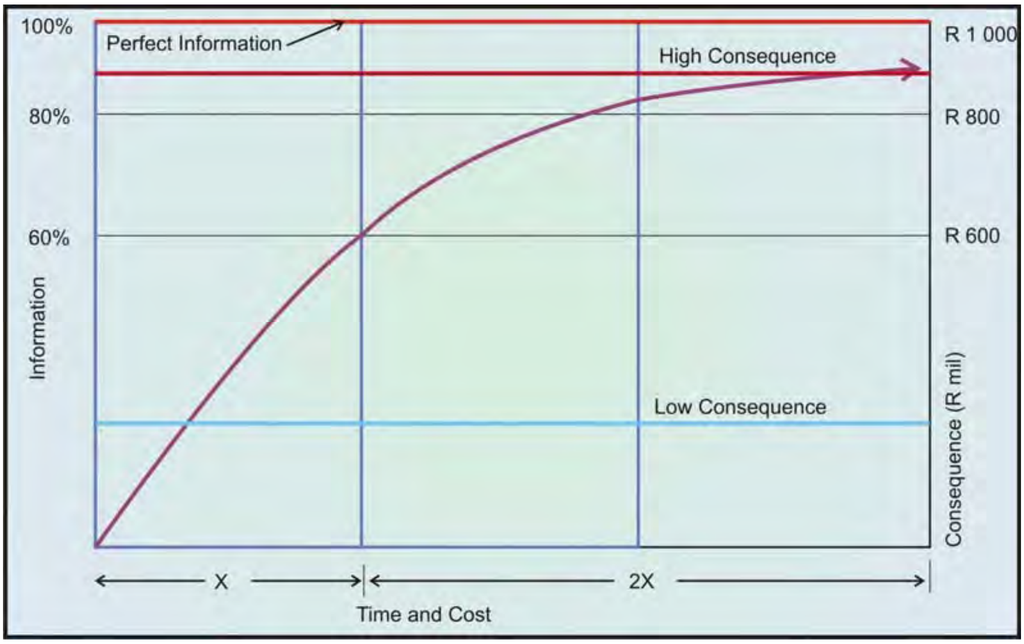


Figure 3-26 Time, cost, information flow and consequence in the decision-making process.

3.6.1 The influence of risk or consequence

In the previous chapter, risk was identified as a function of the probability of failure and the cost of the consequence (Section 2.9.8.5). Risk or consequence is an important consideration that must be included in the decision-making process (Venter, 2010). In high risk programmes, such as the design of a nuclear power plant (NPP) or a major hazardous installation (MHI), the consequence could become the limiting or determining factor for data gathering through site investigations and impact assessments. With a higher consequence, the level of information required for decision-making is also higher and vice versa (Figure 3-26). If e.g. the information level is high at 95%, but the cost of failure is say R 10 billion and a probability of failure is 0.1%, then the risk is R 10 million. In programmes like these, the level of data gathering should be matched by the level of risk. Consequently if the risk is low, then the level of data gathering should be lower, in line with the consequence level.

3.7 The role of assumptions

In the absence of perfect information, *assumptions*[34] must be made. Negative comments or considerations on assumptions in numerous projects lead to the investigation of the role of assumptions in the decision-making process. The negative view on assumptions is based on an expectation that it can be replaced by (perfect) data. In reality, assumptions are also part of the data collection and interpretation process (Figure 3-27). For example, the derivation of aquifer parameters (section 4.3) are based on analytical and numerical analysis techniques or models that make assumptions in order to arrive at solutions. The well known Theis or Cooper-Jacob assumptions are based on aquifers that are limited to two-dimensional, horizontal flow, uniform in thickness, homogeneous and of infinite extent (Kruseman and De Ridder, 1991). Except for the three directly measurable data parameters in groundwater (water levels, abstraction rates and water quality), assumptions form an integral part of the data interpretation and decision-making process. In a decision-making process, assumptions are used to substitute information.

It could be used because there is not sufficient time and budget available to obtain the information or it is not possible to make a conclusive argument based on the information. In many cases, the information is impossible to obtain. It is e.g. not possible to determine exactly all the underground fracture zones and preferential pathways in an aquifer. Assumptions are replaced by information as the decision-making process develops by spending time and money (Figure 3-27).

It is proposed that in the absence of scientific information, conservative assumptions should be used, which is in line with the precautionary principle (Section 2.7). Assumptions used in this way would always have the effect that e.g. more water is available for environmental use than the assumed case. In the case of underground mine flooding evaluations, the assumption would be to determine the upper bound of inflow. Hence to be conservative, assumptions may in the same investigation seem to be contradictory.

[34] A belief or feeling that something is true or will happen, although there is no proof (Oxford English Dictionary, 2006). A statement that is used as a premise of a particular argument that may not be otherwise accepted (Collins English Dictionary, 2006).

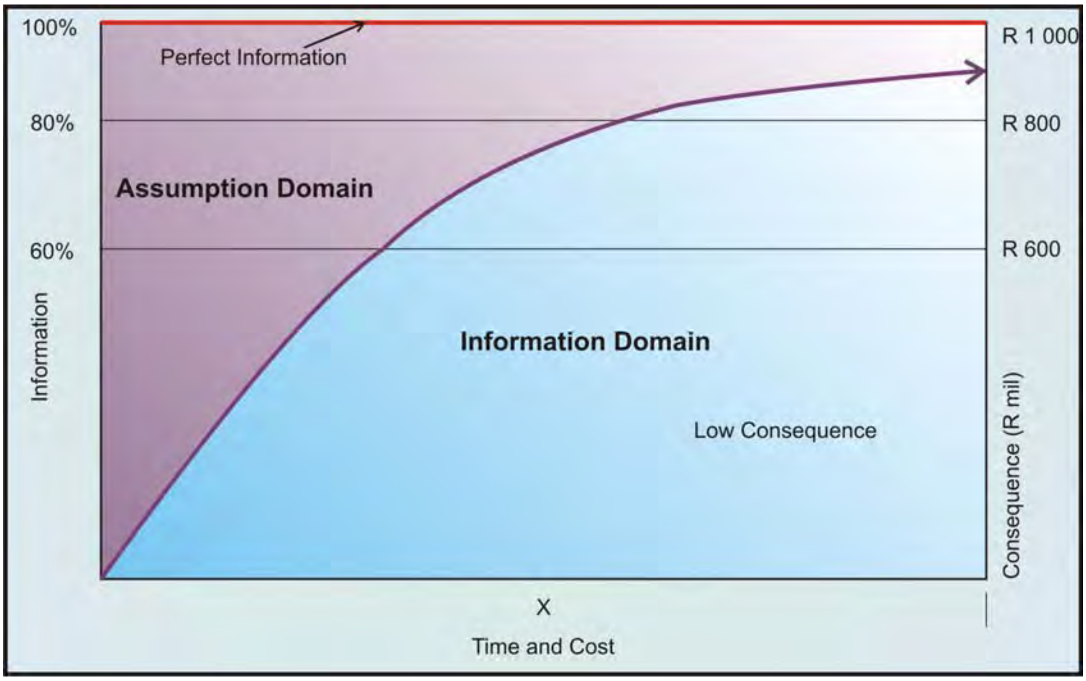


Figure 3-27 Assumption domain and information domain in the decision-making process.

In the case of determining the groundwater component of the reserve, where it is e.g. not known whether wetlands in an area are groundwater supported, the assumption should be made that it is groundwater supported until it can be proven otherwise. This is based on the judicial precautionary approach to decision-making (Section 2.7). Water that is allocated in this way should be considered to be allocated to a *trust account*. If a developer would want to utilize a component of this water and can prove via a detailed geohydrological study that e.g. none of the wetlands are groundwater supported, this component of the water can be *unlocked* from the trust account (Vivier, et al. 2009). Water volumes provided in this way will then represent a *minimum*, which can be increased as more information becomes available that can replace some of the assumptions. Assumptions should therefore be understood as part of the decision-making process. It serves as a safety factor against uncertainty, the higher the uncertainty, the larger the safety factor that is required in the making of assumptions.

3.8 Discussion

The role of *data* and *information* was evaluated on the decision-making process. The aim was to characterise the decision process based on the influence that data and information has on it.

The decision-making process is based on the availability of information. The General Definition of Information (GDI) states that information consists of data + meaning. Information is defined in this section as the accumulation of meaningful data. Information therefore, forms the foundation of decision-making. When data are analyzed, it becomes information which upon interpretation increases the level of knowledge and understanding that is used to base management decisions on. The role that data plays in the information process is similar to the role of pixels in building a picture. A picture cannot exist without pixels, but an accumulation of pixels does not necessarily provide a picture.

To determine the effect that data and information has on the decision-making process, data from three field sites were evaluated. The depth to groundwater level was used as the required variable on which information was required. The change in the average depth to groundwater level was evaluated against the increase in the number of data points. Information was gained based on the change that a data point provides against the previous value. If the average depth to groundwater level after e.g. 3 data points is 8 m and after e.g. the fifth data point is 10 m, then the fifth data point provided information as there was a change. This process was done for all 715 data points collected for the Middelburg Site. It indicated that the information process is logarithmic. The percentage error in the average becomes small at <5% after 30 points and very small at 0.6% after 100 data points and eventually 0.08% after 715 data points. Although it approaches zero, it is asymptotic and will never reach it. *Perfect data* (does not exist, but was used in the same way that an *ideal gas* is defined in physics) was defined as data with a standard deviation of zero. In the case of perfect data, perfect information can be obtained by using one data point. *Perfect information* would therefore amount to 100 % certainty and would idealistically allow the decision-maker to make the *perfect decision*.

The analysis showed that the decision-making process has a logarithmic nature which means that less and less information becomes available with more data. It is similar to the *law of diminishing* *returns* that is used in economics. More data does not necessarily mean more information and is not necessarily better for decision-making. The behaviour of the information in the decision process is also similar to the geostatistical semi-variogram's spherical model that is used to determine the maximum distance between data points that can be used for interpolation.

The value of data was assessed in a data worth or data cost evaluation. The information gained and cost can be optimised by evaluating it against the number of data points. The *optimal* number of data points to gain information based on costs for the Middelburg Site is at 149 data points and at a cost of R29 600. The gathering of more data points would result in a higher cost relative to the information gained. *Sufficient information* could be determined by the analyst at an error value of e.g. <5%, which is at a lower number of data points, which was 30 in the case study. The cost of the sufficient information after 30 data points is R6 000, which is substantially less than the R29 600 of the cost of the optimal no of data points. More data is not better from a sustainability perspective that includes the impact on cost. There is firstly a point of *sufficient information* that should be determined by the analyst which can be increased to *optimal information* in an iterative process, if justified economically.

The effect of risk or consequence is important in the information and decision-making process. With a higher consequence, the level of information required for decision-making is also higher and vice versa

Assumptions form an integral part of the data interpretation and decision-making process. In a decision-making process, assumptions are used to substitute information. It is used, because there is not sufficient time and budget available to obtain the information or it is impossible to make a conclusive argument based on the information at hand. It is proposed that conservative assumptions be used, in line with the precautionary principle.

CHAPTER 4

"Experience is something you get just after you needed it" Unknown

4 THE CHARACTERISTICS OF GROUNDWATER DATA AND IMPLICATION ON DECISION-MAKING IN ENVIRONMENTAL WATER RESERVE DETERMINATIONS

4.1 Introduction

With the proclamation of the National Water Act (Act 36 of 1998) in South Africa, the principle of *the (water) reserve*[35] was introduced. The reserve can be defined as the social and environmental water requirements that need to be allocated before other water uses can be considered. A particular problem in determining the groundwater component of the reserve, is that data is always sparse and associated with a high degree of uncertainty. Studies that have been done to gather and interpret groundwater data usually identify more gaps than provide answers. In a number of studies, it is concluded that data was *insufficient*. The process seems to spiral outwards and produce diffuse answers (i.e. more questions than before the study started) rather than to converge and provide meaningful answers. In this chapter, the nature and characteristics of groundwater data is considered, with specific reference for use in the decision-making process of *reserve determinations*. The following questions on groundwater data characteristics are considered:

1. How is uncertainty in groundwater parameters, data and methods addressed?
2. Which groundwater data and parameters can be used in the judicial decision-making process and how?
3. How is data scarcity addressed?
4. How should assumptions be used and what assumptions are acceptable or applicable?

[35] The Reserve is known as Resource Directed Measures (RDM).

5. Should quality control and quality assurance be taken into account? If so how?

6. How can it be used in the decision-making process for the purposes of the reserve[36]?

Information box 4-A: Excerpt from the National Water Act (Act 36 of 1998)

Part 3: The Reserve

Part 3 deals with the Reserve, which consists of two parts - the basic human needs reserve and the ecological reserve. The basic human needs reserve provides for the essential needs of individuals served by the water resource in question and includes water for drinking, for food preparation and for personal hygiene. The ecological reserve relates to the water required to protect the aquatic ecosystems of the water resource. The Reserve refers to both the quantity and quality of the water in the resource, and will vary depending on the class of the resource. The Minister is required to determine the Reserve for all or part of any significant water resource. If a resource has not yet been classified, a preliminary determination of the Reserve may be made and later superseded by a new one. Once the Reserve is determined for a water resource it is binding in the same way as the class and the resource quality objectives.

Determination of Reserve

16. (1) As soon as reasonably practicable after the class of all or part of a water resource has been determined, the Minister must, by notice in the *Gazette,* determine the Reserve for all or part of that water resource.

(2) A determination of the Reserve must -

(a) be in accordance with the class of the water resource as determined in terms of section 13; and

(b) ensure that adequate allowance is made for each component of the Reserve.

(3) Before determining the Reserve in terms of subsection (1), the Minister must -

(a) publish a notice in the *Gazette* -

(i) setting out the proposed Reserve; and

(ii) inviting written comments to be submitted on the proposed Reserve, specifying an address to which and a date before which comments are to be submitted, which date may not be earlier than 60 days after publication of the notice;

(b) consider what further steps, if any, are appropriate to bring the contents of the notice to the attention of interested persons, and take those steps which the Minister considers to be appropriate; and

(c) consider all comments received on or before the date specified in paragraph *(a)*(ii).

Preliminary determinations of Reserve

17. (1) Until a system for classifying water resources has been prescribed or a class of a water resource has been determined, the Minister -

(a) may, for all or part of a water resource; and

(b) must, before authorising the use of water under section 22(5), make a preliminary determination of the Reserve.

(2) A determination in terms of section 16(1) supersedes a preliminary determination.

[36] It is important to note that the reserve is regulated by legal processes.

Giving effect to Reserve

18. The Minister, the Director-General, an organ of state and a water management institution, must give effect to the Reserve as determined in terms of this Part when exercising any power or performing any duty in terms of this Act.

4.2 The characteristics of groundwater data and information

Since it is known that groundwater data is spatially and temporally variable, suitable analysis methods can be used that deals with uncertainty. Statistical methods are suited to deal with and characterize variability in data so that the analyst is aware of the potential effects of variability and uncertainty (Poeter and McKenna, 1995; Kaplan, 1997).

As discussed earlier (Section 2.9.4), there are two schools of thought in statistics namely, the classical statisticians or frequentists and the Bayesian statisticians. The problem using the frequentist approach, is that it does not provide answers in cases where data is sparse and uncertainty high, which is mostly the case in groundwater problems. For the purposes of groundwater data analysis, it is proposed that the frequentist approach is used only where sufficient data is available. In terms of the decision-making process, the Bayesian approach is recommended with the use of a prior analysis that is updated with a posterior analysis as more data become available (Freeze et al. 1990). The process should be iterative and converges to an acceptable or optimal outcome (Figure 3-25).

4.2.1 Sparse data and uncertainty

The problem with groundwater data scarcity and uncertainty can be addressed by utilizing geospatial and statistical methods that are designed for this purpose (Poeter, 2006). The various methods that are recommended to be used for the purposes of the assessment and interpretation of the groundwater data are:

1. Temporally variable data (rainfall, recharge, water levels etc),
2. mean
3. standard deviation
4. percentile values for levels of assurance

5. regression (correlation)
6. hypothesis testing
7. Bayesian estimation using preconditioning and posterior conditioning.
8. Spatially variable data (transmissivity, rainfall, recharge, water levels etc).
9. Linear interpolation.
10. Semi-variograms.
11. Kriging.
12. Inverse distance to a power.
13. Nearest neighbour etc.

These variables should be used to determine statistical distributions (also known as probability density functions) that can be used to perform forward assessments using analytical and numerical models. Important outcomes are to determine the variability of the various geohydrological parameters as it is the variability (temporal and spatial) that increases the uncertainty, which in turn makes the management of the resource more challenging. If the uncertainty is known, provision can be made for it in the interpretation and decision-making process.

4.3 Uncertainty associated with groundwater parameters and methods of analysis

Geohydrological parameters that are used to interpret and analyze information are evaluated in terms of its uncertainty and impact on decision-making, especially in judicial processes such as the reserve determination for the National Water Act (Act 36 of 1998). Data parameters are listed and classified in terms of the nature of the variable, field measurement methods, statistical distribution, methods of interpretation and level of uncertainty. The level of uncertainty should be accounted for in any decision-making process. The evaluation was done to determine which parameters can be measured directly in the field, which are derived and which are estimated etc.

No	Variable	Variable character	Variable field measurement or assessment method	Distribution	Methods of assessment	Methods of interpretation	Uncertainty in parameter, methods of assessment and interpretation
1	Water levels (piezometric heads)	Spatial and temporal variable	Measured (quantitative)	Probabilistic (spatially variable)	Measured at borehole (data points)	Statistical in terms of temporal variability. Geostatistical in terms of spatial variability (Kriging, inverse distance etc)	Low
2	Abstraction rates	Spatial and temporal variable	Measured (quantitative)	Probabilistic (spatially variable)	Measured at borehole (data points)	Statistical in terms of temporal variability. Geostatistical in terms of spatial variability (Kriging, inverse distance etc)	Low
3	Water quality	Spatial and temporal variable	Measured (quantitative)	Probabilistic (spatially variable)	Measured at borehole (data points)	Statistical in terms of temporal variability. Geostatistical in terms of spatial variability (Kriging, inverse distance etc)	Low
4	Rainfall	Spatially and temporally highly variable	Measured (quantitative)	Probabilistic (normal or log normal)	Measured at rainfall stations	Statistical in terms of temporal variability	Low to medium
						Geostatistical in terms of spatial variability (Kriging, inverse distance etc)	
						Monte Carlo assessments in terms of future probabilities	
5	Hydrogeology	Spatially variable	Mapped (spatially quantified)	Probabilistic (spatially variable)	Measured/verified at borehole (data points)	Geostatistical in terms of spatial variability. Although hydrogeological zones are usually consistent within polygons. This data is usually limited to two dimensions	Medium to high
6	Transmissivity (hydraulic conductivity)	Spatially variable. Non-linear	Derived (quantitative from field tests)	Probabilistic (spatially variable)	Measured at borehole (data points) using aquifer tests	Geostatistical in terms of spatial variability (Kriging, inverse distance etc). Statistical in terms of temporal variability.	Medium to high
7	Other sources and sinks (dams, springs, wetlands, alien vegetation etc)	Spatially variable.	Derived (quantitative from field tests) or estimated using a qualified guess	Probabilistic (spatially variable)	Qualified using field measurements or estimated using remote sensing to enhance the qualified guess	Statistical in terms of temporal variations and geostatistical (Kriging, inverse distance etc) in terms of spatial variations	High
8	Groundwater recharge	Spatial and temporal highly variable	Estimated (with a qualified guess)	Probabilistic (normal or log normal)	Qualified using chloride or CFC methods.	Statistical in terms of temporal variability	Very high
		Non-linear	Can only be qualified using long-term monitoring data		Qualified using isotope analysis	Geostatistical in terms of spatial variability (Kriging, inverse distance etc)	
					Qualified using analytical and methods such as SVF or numerical models	Monte Carlo assessments in terms of future probabilities	
9	Porosity &Storativity (specific storativity, specific yield)	Spatially variable. Non-linear	Estimated (with a qualified guess)	Probabilistic (spatially variable)	Qualified at borehole (data points) using aquifer tests that enhances the qualified guess	Geostatistical in terms of spatial variability (Kriging, inverse distance etc).	Very high
			Can only be qualified using long-term monitoring data				

In most cases, variables such as transmissivity, recharge, porosity and storativity are estimated (qualified guess) and assumed to be spatially consistent within the same hydrogeological unit or formation

Table 4-1 Geohydrological data parameters, assessment methods, interpretation and uncertainty.

The evaluation indicated that (Table 2-2):

1. Direct field measurements: Only three direct groundwater parameters namely; water levels, abstraction rates and water quality, can be quantitatively and directly measured in the field. The level of uncertainty at the data points is low, but spatial analyses techniques are required to interpolate the data across the domain of interest. The spatial interpolation introduces another level of uncertainty.
2. Rainfall is also an important direct field measurement that can be made. It is the driving force behind groundwater recharge and requires statistical analysis. It can be considered as an indirect parameter as it influences recharge that is a more complex variable. Temporal and spatial uncertainty in rainfall is also introduced where interpolations between measurement stations or points must be made.
3. Hydrogeological zones are well mapped across South Africa from geological data sources. This data is generally available for the two-dimensional case. Hydrogeology becomes much more complex when three-dimensional considerations are made that introduces a higher level of uncertainty.
4. Derivations based on analytical models: Hydraulic groundwater (or aquifer) parameters such as transmissivity (or hydraulic conductivity) and storativity are indirectly derived from analytical and numerical models. The data from field tests are analysed to arrive at parameter values. The assumptions related to these analyses techniques and indirect nature of these parameters introduces a high level of uncertainty (Kruseman and de Ridder, 1991).
5. Sources and sinks (dams, springs, wetlands, alien vegetation etc) that are estimated using qualitative field measurements or remote sensing. Remote sensing would e.g. be used to determine the area of a wetland or alien vegetation patch which will be used to derive water use values. Water use values of e.g. tree plantations per ha is known. A higher level of uncertainty is introduced as derivations are made based on remote sensing or GIS information. The quantification of these sources and sinks are usually conservative estimates as very few field measurements are available.
6. Estimations based on a qualified guess: The groundwater parameters of recharge, porosity and storativity cannot be measured or determined directly from field

measurements such as aquifer tests. In most cases only an initial estimation or guess can be made that can be qualified or quantified, using analytical techniques (chloride, CFC, isotope methods etc) or from long-term monitoring data. If field samples are taken, there is still a spatial variability. The conundrum is that these two parameters are the most important in determining sustainable resource quantities.

Of the nine basic groundwater parameters (excluding rainfall), only three, (water levels, abstraction rates and water quality) can be measured directly in the field, with the hydraulic parameters relying on derivations from field tests. Recharge and storativity are estimations based on a qualified guess. Geohydrology can in this sense be classified as a *non-unique science* associated with a *high degree of uncertainty*. The aim of data gathering programmes should be to obtain *sufficient* or *optimal* data and not to pursue exact data (Section 3.3.2). The data should then be interpreted to characterize and understand the parameters that are required for decision-making. The level of uncertainty needs to be reduced for the purposes of decision-making and for management purposes as it becomes very difficult, expensive and even impossible to reduce uncertainty beyond certain ranges (Section 3.6). Data and uncertainty can therefore not be evaluated in the absence of a decision-making framework or methodology (Van Blerk, 2000), which should be used to determine when data is sufficient.

4.4 The effect of scale on data and information

Scale (temporal and spatial) is one of the most important aspects when considering data and uncertainty. If e.g. one data point represented by one borehole in an aquifer is considered, which abstracts water at a known rate with a known water level, it can be considered as close to a deterministic (i.e. unique) situation as possible. In reality the water level will change with time as recharge driven by rainfall varies over time. As more boreholes are considered that are distributed across an aquifer with various abstraction rates and water levels over time, the problem becomes a variable and statistical one. Scale therefore influences uncertainty and in general, the larger the scale (or smaller beyond a certain cut off), the larger the uncertainty. An important requirement to assess in the objective of any study is the scale and resolution of the assessment.

This is where the concept of a representative elementary volume (REV) becomes important (Bear, 1979; Botha; 1996) (Figure 4-1). It means that a given groundwater problem cannot be evaluated below or above a certain scale. If e.g. a thin section of a sandstone is taken to determine the porosity, and the sand grains are 1-2 mm in diameter, then a sample in a 10 mm square area could either lead to the conclusion that the aquifer is impermeable with 0% porosity if taken at point 1 or that the aquifer is infinitely permeable with a 100% porosity of taken at point 2 (Figure 4-1). The REV for this sandstone would e.g. be a sample size of a 200 mm square, to obtain the representative or average effect. If an example of a dolomite is used, a pump test done in one area could conclude that the dolomite is highly permeable or vice versa, which could deviate significantly from the average. What is required, is e.g. several pump tests across a larger area with statistical analysis, to correctly interpret and understand the hydraulic properties of the dolomite.

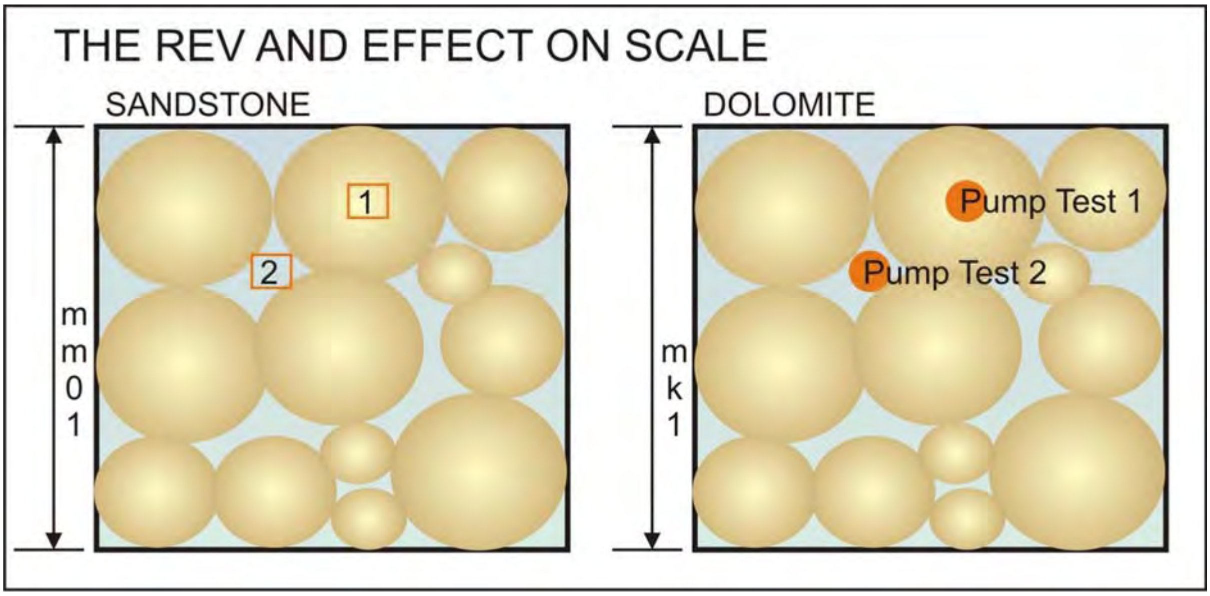


Figure 4-1 Schematic representation of the REV and effect of scale.

The same effect is valid for groundwater assessments in the reserve determination on regional catchment scales. Groundwater assessments are usually done on well field scale, which is typically not larger than 3 km x 3 km. If data is gathered on this scale, it could not be used on a quaternary catchment scale, which is usually much larger. The average scale of the quaternary catchment e.g. at the Outeniqua Study is 162 km^2 (13 km x 13 km), (AGES, 2010b).

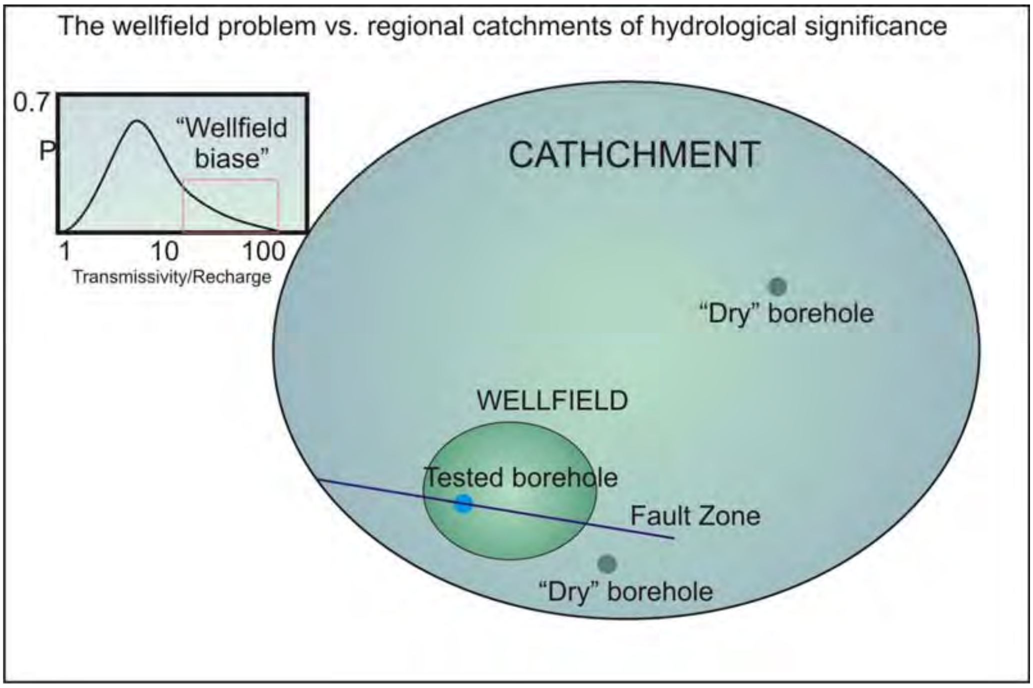


Figure 4-2 Schematic representation of the effect of scale on quaternary vs well field approaches.

In hydrogeological investigations, structural geological and geophysical surveys are used to focus borehole development in the high permeability zones. Boreholes that are drilled in *dry* areas are not used for e.g. in pump test analysis or recharge estimations. The result is that the high end of the range for transmissivity or recharge, which usually follows a skewed distribution is determined (Figure 4-2). Data gathered in this way does not reflect the average hydraulic status of the regional aquifer, but only that of the permeable well field. Groundwater models will typically calibrate on the average parameters (e.g. transmissivity) if regional water levels are used, which is usually lower than the values determined from pump tests on well field scales. The over estimation of regional aquifer parameters based on local assessments, can be termed *well field bias*.

This is a common problem where *up-scaling* of the investigation area should be done with *down-scaling* of the recharge and other aquifer parameters. This error of up-scaling in the investigation area, using the same aquifer parameters and recharge cause a gross overestimation of e.g. groundwater recharge on a regional scale, as will be shown later (AGES, 2010b).

The analyst must pre-determine the scale of the assessment as it will influence the data collection program and the uncertainty. All data and uncertainty will be relative to the scale of the assessment. The analyst should adapt the methodology and interpretation of results to the scale of the assessment and/or integrate the outcomes at different scales to make provision for scale effects. If for example all boreholes are drilled only on pre-determined fault zones, then the more data that is collected would lead to more skewed parameter values and hence less accurate assessment of the problem.

4.5 Quality control

For the purposes of groundwater studies, particularly related to the reserve, a quality control and quality assurance system will be required to ensure that data, information sources and assumptions are listed and referenced. The reason for collection of data must be to aid in the decision-making process and not for the sake of collecting data. A checklist is proposed of which Table 4-2 is an example. Important aspects that should be checked are data adequacy and levels of uncertainty.

Table 4-2 Geohydrological data parameters, and reference list (Vivier et al. 2009).

No	Data type	Source and assumptions	Date
1	Catchment area	WRMS 2000 quaternary catchment areas	2006
2	Rainfall	Data received for all rainfall stations in the study area from SA Weather Bureau	2009
3	Recharge	The same recharge % values as in Parsons R. (2006) Outeniqua Coast Water Situation Study. Report to the Department of Water and Environmental Affairs - were used as an initial guide. The recharge were not determined from MAP but from an assurance level. An assumption was made that recharge is lower, based on the scale of the assessment.	2009
4	Dam seepage	DWA :National Water Resource Planning. Outeniqua Coast Water Situation Study. Appendix 2. Dam areas were digitized that were visible from Google Earth satellite images	2006
5	Geology (primary)	1:250 000 Geological map and GIS geological delineations from Parsons (2006)	
6	Geology (primary) area (km^2)	1:250 000 Geological map and GIS geological delineations from Parsons (2006)	2008
7	Depth to water level (m)	Determined as an average per quaternary catchment, based on NGDB borehole data and borehole data received from Parsons in Wish Excel format.	2008
8	Min depth to water level (m)	Assumed to be 5 m from surface as an average across the quaternary catchments	2009
9	Max aquifer depth (m)	Assumed to be 110 m depth as deep groundwater is not evaluated in this assessment	2009

No	Data type	Source and assumptions	Date
10	Water level management constraint (m)	Assumed to be 25% of the saturated thickness. Decline of water levels lower than a depth representing 25% of the saturated thickness is not allowed.	
11	Aquifer storativity (1)	Assumed to be 0.001 as an average. Due to the fact that transient stochastic simulations were not done and that aquifer storativity values was not directly determined/estimated from pump tests, it was conservatively assumed to be on the low end.	
12	Groundwater volume in storage (m^3)	Calculated as total groundwater volume in storage	
13	Max usable groundwater volume in storage (m3)	Calculated as usable groundwater volume in storage - up to the management constraint	
14	MAP (mm/a)	Calculated from rainfall station data	
15	MAE (mm/a)	Calculated from the George Weather Station A pan potential evaporation data	2009
16	MAR (%)	Data used from EWR studies (Sherman Consulting and Southern Waters)	2009
17	MAR (m3/a)	Data used from EWR studies (Sherman Consulting and Southern Waters)	2009
18	MAR (mm/a)	Calculated	
19	Rainfall 95% assurance (mm/a)	Calculated per quaternary catchment from rainfall records	
20	Recharge 95% (m3/a)	Calculated	
21	Number of dams	DWEA :National Water Resource Planning. Outeniqua Coast Water Situation Study. Appendix 2. Dams and areas were digitized that were visible from Google Earth satellite images and GIS maps	2006
22	Avg seepage (mm/a)	Conservatively assumed to be 100 mm/a.	
23	Dam Seepage Area (km^2)	Digitised from Google Earth and Geographic Information(GIS) maps for all the existing dams.	2008
24	General autho- rizations	Data from Parsons R. (2006) Outeniqua Coast Water Situation Study.	
25	Number of abstraction boreholes (Other)	Borehole abstraction data from Parsons R. (2006) Outeniqua Coast Water Situation Study with updates based on NGDB data.	2006
26	Total borehole abstraction (m^3/a)	Same as above	2006
27	Number of livestock farms	Number of farms were determined from GIS data and maps. It was assumed that each farm uses 0.25 ℓ/s for 24 h/d for livestock water	2008
28	Total livestock farm usage (m^3/a)	Calculated from the number of farms and assumption of 0.25 ℓ/s per farm for 24 h/d	
29	Number of mines	There are no mines in the study area	
30	No of comm- unities	Determined from GIS base maps	
31	People in community	Data per community from Stats SA	
32	Total community borehole usage (m^3/d)	Allocated at 25 ℓ/person/day	
33	Average Farm irrigation area (ha)	DWEA :National Water Resource Planning. Outeniqua Coast Water Situation Study. It was assumed that between 1-5% of irrigation uses groundwater. If values of greater than 1-5% were used, it would use more than the recharge, which is not possible	2006

No	Data type	Source and assumptions	Date
34	Average Forestry area (ha)	DWEA :National Water Resource Planning. Outeniqua Coast Water Situation Study. Appendix 8. It was assumed that 2% of the forestry area tap groundwater at a rate of 1200 mm/a. If values of greater than 2% were used, it would use more than the recharge, which is not possible	2006
35	Average Riparian Alien veg (ha)	DWEA :National Water Resource Planning. Outeniqua Coast Water Situation Study. Appendix 8. It was assumed that 5% of the riparian veg area tap groundwater at a rate of 1000 mm/a. If values of greater than 5% were used, it would use more than the recharge, which is not possible	2006
36	Wetlands (Ground water) (km^2)	Source: M Rountree (2008): The wetland shape file has been generated predominantly from the SANBI wetland probability map. Map Digitized and Google Earth Map and GIS. The max groundwater component was determined from the model at 10-20% of the total potential wetland water use. Values higher than these could not be explained by the groundwater volumes. The wetland water use was estimated at 1000 mm/a.	2008
37	No of springs	Mosselbaai :Data CD and GISCOE Map Digitized and Google Earth Map and Geographic Information Systems (GIS)	2006
38	Spring flow (m^3/a)	Assumed at 0.5 ℓ/s per spring	
39	Evapo-transpiration flow loss 1 (m^3/a) % of catchment	Calculated as a % of the catchment for model to calibrate on surface water base flow values. It was assumed that the groundwater component of baseflow was 70% of the total base flow.	
40	Net GW base flow (m^3/a)	Base flow values were obtained from BKS (Estelle V Niekerk) for drought low flow values. Values were generated from Spatsim and was in line with the lower end of recharge used to cater for drought conditions	2007
41	Net Base Flow Required by EWR - Drought low flows (m^3/a)	The EWR values were obtained from SouthernWaters (Dr Brown) and Sherman Consulting (Dr Sherman)	2009

4.6 Method statement for data collection and interpretation

Data collection and interpretation without a framework within which it can or should take place would lead to ineffective and expensive exercises. A formal step wise methodology is proposed that would ensure uniformity in the data collection and analysis approach (Figure 4-3):

1. Define the problem (groundwater reserve level), objectives and outcomes in terms of the decisions that will have to be taken.
2. Plan the data collection program with interpretation and the scale of accuracy in the assessment. The scale and resolution of the assessment is important. It is recommended to start at a large (Primary Catchment) scale with quaternary catchment scale resolution for rapid and intermediate reserve determinations that can be stepped up in areas where

more detail is required. It is not advisable to start at a wellfield scale as the data accuracy would not be met at this scale.

3. Obtain existing data from data libraries (such as the GRDM library that must still be created). Basic data (catchment sizes, hydrogeological units, rainfall, recharge estimates, abstraction rates etc) for all catchments and hydrogeological units exist for the whole country.

4. Interpret existing data using deterministic and statistical methods. Determine the variability of the data and characterize the uncertainty in parameters.

5. Resource quantification (modelling) should be done to determine the quantity of water available for the areas considered. This is important to determine the interactions between components such as boreholes and wetlands or rivers, and to determine sensitive parameters.

6. Identify data gaps. This is required to focus the field work which is often expensive. If from the resource assessment, it is indicated that e.g. wetlands could have a more important influence on groundwater, the focus of data collection should not only be on boreholes.

7. Only at this stage should field work programs be done that are focused at collection of the most important data parameters. From the statistical analysis (variability and uncertainty assessment) and a data worth assessment, data points or areas that require more data can be identified. This action is foreseen for intermediate and especially comprehensive reserve levels.

8. Data re-interpretation and characterization of uncertainty, using added value of field work component. Based on the outcome, additional re-focused field work and data collection could be done to decrease the level of uncertainty.

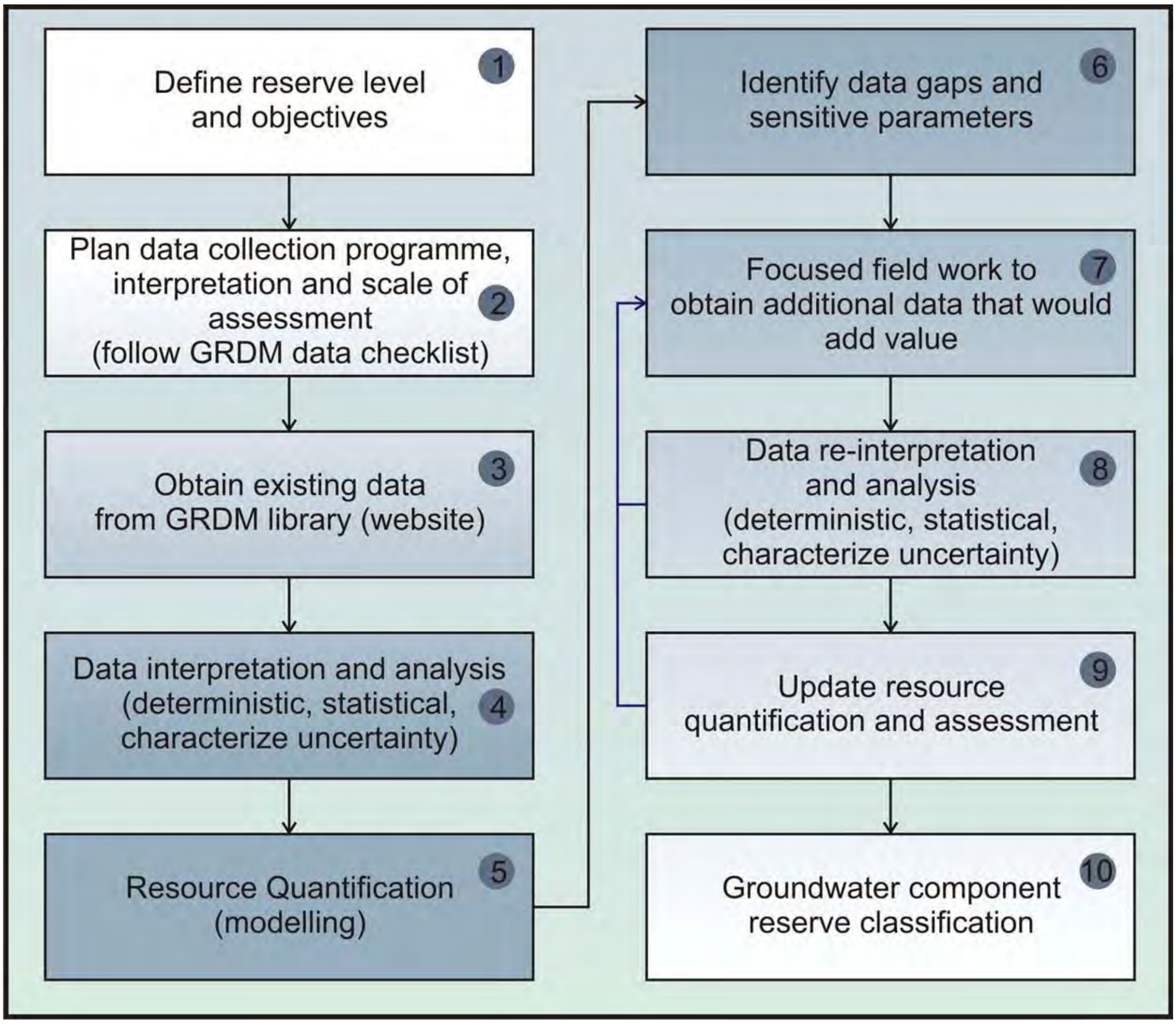


Figure 4-3 Proposed data evaluation and collection methodology.

9. Update resource quantification assessment. Based on the outcome, additional re-focused field work and data collection could be done to decrease the level of uncertainty. The field work programmes would add value as based on the model output where only the (most) important parameters can be focused on that will add value in terms of the decision-making process.
10. Classification of the groundwater component of the reserve.

The data collection, interpretation and resource quantification are linked towards the decision-making or output of the GRDM. The data collection cannot stand alone as it would be without context. Data collection, especially field work, is associated with high expenses and is limited by financial constraints. Statistical methods and analysis techniques such as modelling should be used to optimize data collection programmes.

4.7 Discussion

Geohydrology is not an exact science and there are no unique data sets or solutions. It is therefore not useful to consider data as exact and to aim at obtaining unique solutions. Data forms the basis of the decision-making process and if it is accepted that data is sparse and uncertain, then the approach to the problem, analysis techniques and decision-making process can be adapted to the nature of the problem and the decision outcomes. When data is analyzed, it becomes information which upon interpretation, is used to base management decisions on. The aim and purpose of groundwater data collection for the reserve must be defined before the process is actually started. Decision-making for the purposes of the reserve is driven by judicial and principle based considerations. The purpose of data collection for the reserve must be to assist in the quantification of the resource for the purposes of the reserve determination.

Evaluation of the data and uncertainty showed that groundwater data is highly variable, sparse and associated with a high degree of uncertainty. Of the nine groundwater parameters that are usually gathered, only three (water levels, abstraction rates and water quality) can be measured directly in the field. The hydraulic parameters (transmissivity) are derived from field tests and very important parameters such as recharge and storativity are estimations or qualified guesses at best.

The data collection and interpretation process should be defined within the constraints of the parameters which should be based on the decision-making framework. A formal, step-wise methodology for data collection, interpretation, and field work is proposed. More data does not necessarily mean better decision-making. The analyst should adapt the methodology and interpretation of results to the scale of the assessment and/or integrate the outcomes at different scales to make provision for scale effects. If for example all boreholes are drilled only on pre-determined fault zones, then the more data that is collected would lead to more skewed parameter values and hence less accurate assessment of the problem. *Sufficient* and/or *optimal* data equates to better decision-making.

Data collection, interpretation and resource quantification assessments must be linked towards the decision-making or output of the GRDM. The data collection cannot stand alone as it would be without context, meaning and value. Data has value and the value of data decreases with an increase in data points. Data collection, especially field work, is associated with high expenses and is limited by financial constraints. Statistical methods and analysis techniques such as modelling should be used to suffice and/or optimize data collection programmes.

CHAPTER 5

"All models are wrong, but some are useful" George Box

5 A SYSTEMS MODELLING APPROACH TO ENVIRONMENTAL DECISION-MAKING

5.1 Introduction

In the previous section (Section 2), it was established that the decision-making process should be based on a holistic approach that integrates the specialist and individualist (or reductionist) components with the overall strategy. It was found that the decision-making process converges with an increasing number of data points and that the value of data as information reduces along this process (Section 3). Groundwater and environmental data is by nature sparse and associated with a high degree of uncertainty (Section 3). A tool is required that can interpret the data to turn it into a higher level of information to assist the analyst *to see beyond the horizon*. That tool is established in science and engineering as systems modelling that makes use of analytical or mathematical techniques to quantify environmental problems (Section 2.9.8).

5.2 Why intuitive decision-making is misleading

In a complex world, we cannot rely anymore on intuitive decision-making. To illustrate this point, a number of technical people were asked to provide an answer to the following thought experiment (Gladwell, 2000)[37]:

Suppose we can take a piece of paper and fold it double and again and repeat the folding 50 times. What would the height of the resulting paper stack be? The answers from the various people ranged from 3 cm to 50 km (Table 5-1). The real answer is more than the distance to the sun and back. The reason for this is that the problem is exponential in nature and we as human beings are used to solving linear problems (Section 2.4). Where we cannot trust our intuition, a simple spreadsheet calculation shows that after 1024 folds, the distance is 8.98+298 km. The spreadsheet program cannot calculate beyond this number. The analyst must therefore make use of analytical and mathematical techniques to eliminate intuitive biases.

Table 5-1 Answers provided by a sample on the thought experiment.

No	Person	Answer	Unit
1	Stephan	10	cm
2	Reuben	50000	m
3	Corne	3	cm
4	JC	900	m
5	George	1	m
6	Andre	1.1	m

"The world – as much as we want to – does not correspond with our intuition." M. Gladwell

5.3 Systems thinking and theory

The rate of change in economical, technological, social and environmental spheres is occurring at an increasing rate, while the complexity is growing. This challenges decision-makers to adapt and find new ways to deal with problem solving in an ever changing environment. *Systems*[38] *thinking* is the understanding of how components in complex systems influence each other as a whole (Sterman, 2000; Sterman, 2002). System thinking can be regarded as viewing problems as parts of a system rather than considering specific parts of outcomes of individual components. It is increasingly used in decisions regarding sustainable development (Nooteboom, 2007). Systems theory is defined as the transdisciplinary study of the organization of phenomena, independent of their type, substance, or spatial and temporal scale of existence. It investigates the principles common to all complex entities and the conceptual and mathematical models that can be used to describe them (http://pespmc1.vub.ac.be/SYSTHEOR.html).

There are essentially two views of general problem solving. The historical view is that of cause and effect or event-orientated and the more complex systems view that consists of components connected via interaction pathways with feedback loops (Figure 5-1). If we use a business example, then a business might aim to obtain a bigger market share by e.g. cutting prices. The effect is that sales rise with success, which was the purpose of the event orientated view. The side effect or feedback loop is however such that competitors react and cut prices and sales drop

[37] This thought experiment was taken from Gladwell (2000) and redone in the office with six people.

[38] A system is defined as a group of things that are connected to work together (Oxford English Dictionary, 2006). A group or combination of interrelated, interdependent, or interacting elements forming a collective entity (Collins English Dictionary, 2006).

again. It resulted in short-term success and yesterday's solution becomes today's problem (Sterman, 2000). The problem with historical viewpoints is that secondary problems could be created in the aim at solving the current problem that could result in bigger downstream problems than the original one. This is a classical example for the reason of the existence of most environmental problems that we face today.

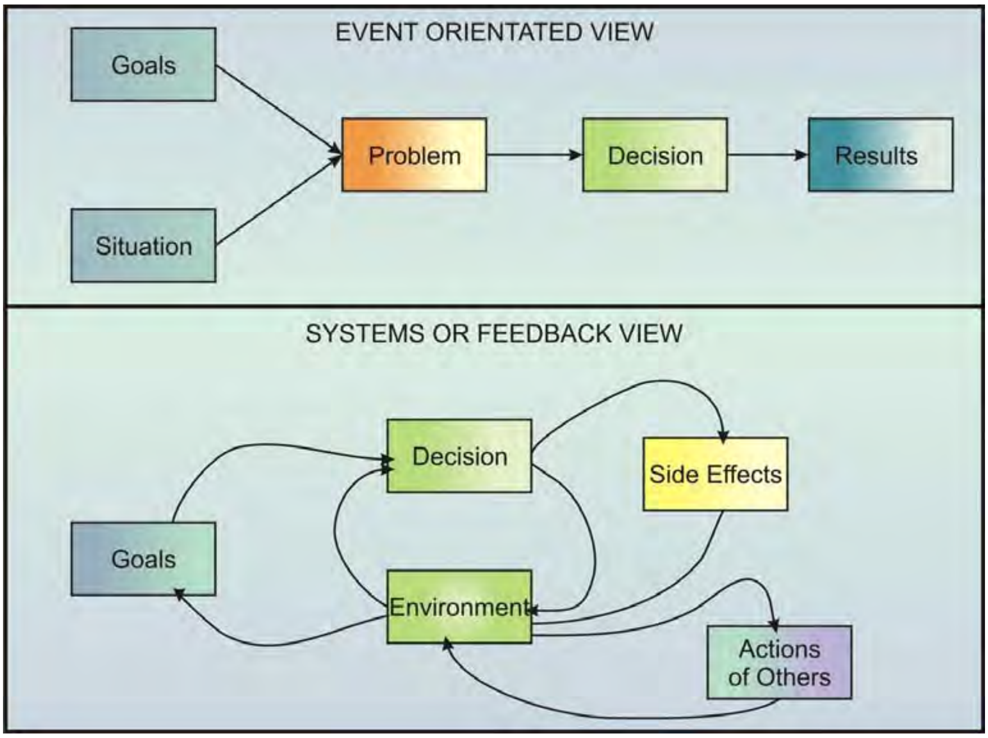


Figure 5-1 Schematic representation of the event-orientated and systems or feedback views.

Systems can be open and linear or closed and cyclic (Figure 5-2). A closed system adheres to the principle of conservation of mass and energy but in an open system, mass and energy can enter and leave the system. An open system is e.g. a highway where cars can enter or leave, and a closed system is e.g. the hydrologic cycle (Figure 5-4). Most of the systems that will be considered here would be based on the environment, such as groundwater that is closed. Anthropogenic interaction is very important and could have an important influence on natural systems. A flow system consists of components that interact with each other via pathways which represent processes along which the flow of mass and energy takes place (Strahler and Strahler, 2005).

In nature, a complex system could be represented by e.g. eco-systems. An amphibian specialist will e.g. consider the information with regard to toads in a wetland (i.e. reductionist approach) where a systems approach would be to consider the interaction of the toad with other fauna, flora, air, soil water etc, in and outside the wetland (i.e. holistic and systems approach).

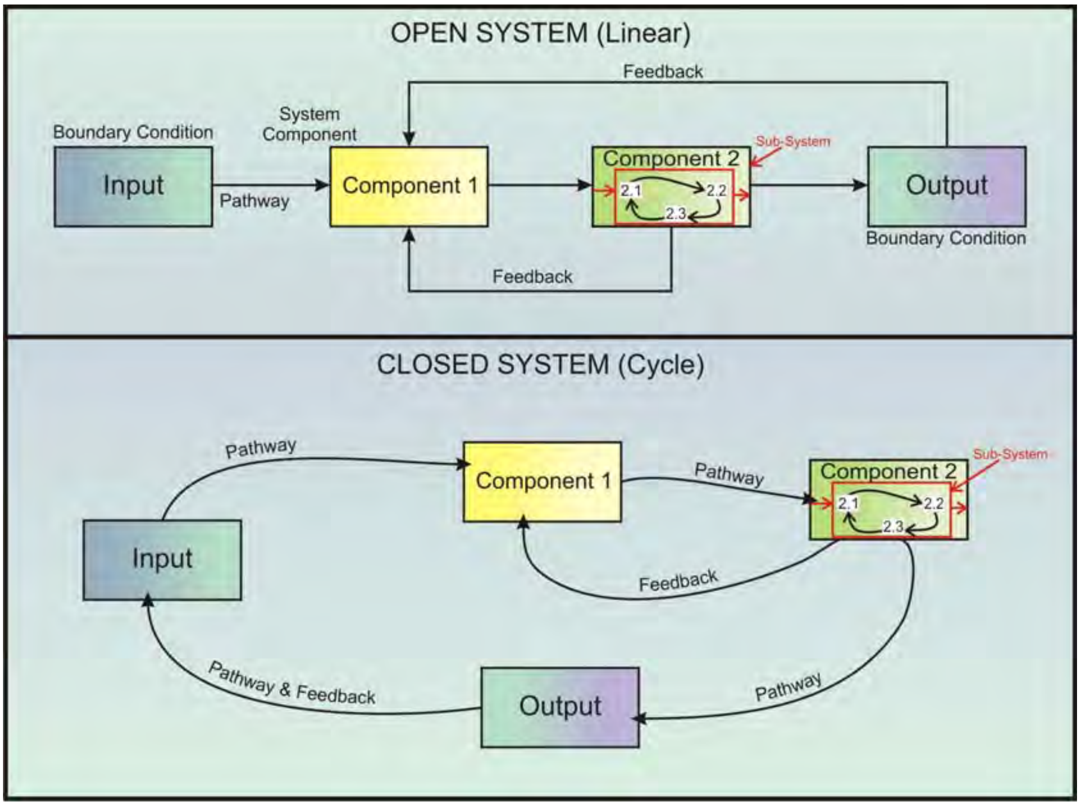


Figure 5-2 Schematic representation of linear and cyclic systems with sub-systems.

5.3.1 Systems models

Models are widely used in systems assessments (Sterman, 2000). The use of systems models have expanded to include the intangible effects, such as political influences and policy making. It can be used to include the influence of legal process and socio-economic effects and is therefore ideal for use in environmental decision-making. Systems models are used to simulate linear and non-linear effects between and in system components. In a system, the relationships and interactions between components are as important as the components itself. The approach to problems is cyclical rather than linear[39]. If problems are solved in a linear cause and effect way,

[39] (http://pespmc1.vub.ac.be/SYSTHEOR.html)

it could create secondary problems that are worse than the original problem. The use of systems thinking and systems models aim at reducing the risk of secondary problems by following a holistic approach (Sterman 2002; Szczerbicki, 2004). There are numerous types of systems that range from biological systems (e.g. the brain or heart), mechanical systems (e.g. car), ecological systems (e.g. predator-prey) and social systems (groups or a town). Complex systems usually have sub-systems that can be evaluated in more detail as a sub-routine (http://managementhelp.org/systems/systems.htm; Figure 5-2).

Systems have positive and negative feedback loops. Positive feedback does not mean that it has advantages and vice versa. Positive feedback reinforces the trend while negative feedback inhibits a trend (Sterman, 2000). Systems are defined by elements, relationships, wholes and rules.

5.4 Sustainability from a systems viewpoint

If sustainability (Section 2.5) of e.g. mining is considered, then there is a popular view that mining is not sustainable as the resource is depleted with time. From a systems viewpoint, mining could have sustainable downstream effects such as leaving behind a city like Johannesburg that drives the economy of a country and a continent (Figure 5-3). The pre-requisite in this case is that the downstream effect must reach critical mass and grow beyond it before the primary activity ends. From a systems viewpoint, there is a more complex answer to the question of sustainability than merely the depletion of a resource. To illustrate this, if the case of mining at e.g. Koffiefontein can be considered, it could be viewed as a non-sustainable development. It is well known that when mining stopped in the early 2000's, the town became almost desolate. Questions that have to be asked are:

- What was the value of people that were trained at the mine and now work elsewhere?
- What was the value of taxes paid to government to develop the country?
- What is the cost of environmental liabilities? Now and into the future?

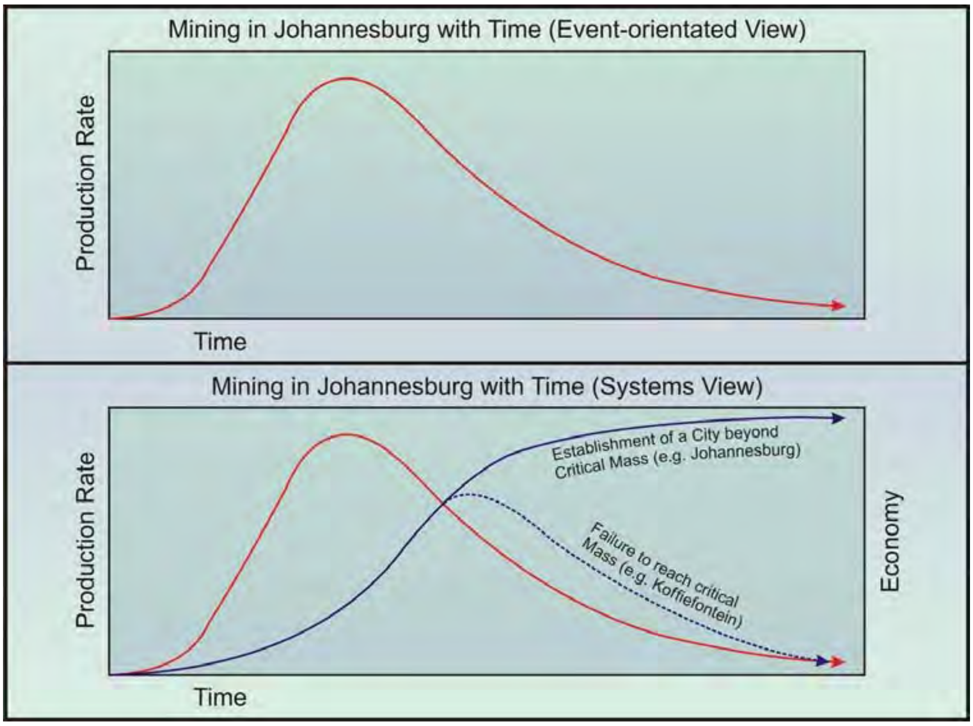


Figure 5-3 Sustainability from a systems viewpoint.

From a systems viewpoint, environmental protection or degradation and sustainability is a more complex aspect than if it is considered linearly. Mining could e.g. spawn a city which raises the standard of living of people that again raises the demand for gold which in turn keeps mining profitable. The secondary or downstream effects from the primary activity must be considered, which could increase or decrease the level of sustainability of activities.

5.5 A systems approach to the understanding and modelling of complex systems

In a complex world, models (Section 2.9.8) and in particular systems models are more widely used for problem solving. It is important to determine the philosophy behind systems modelling and the modelling process for the purposes of decision-making.

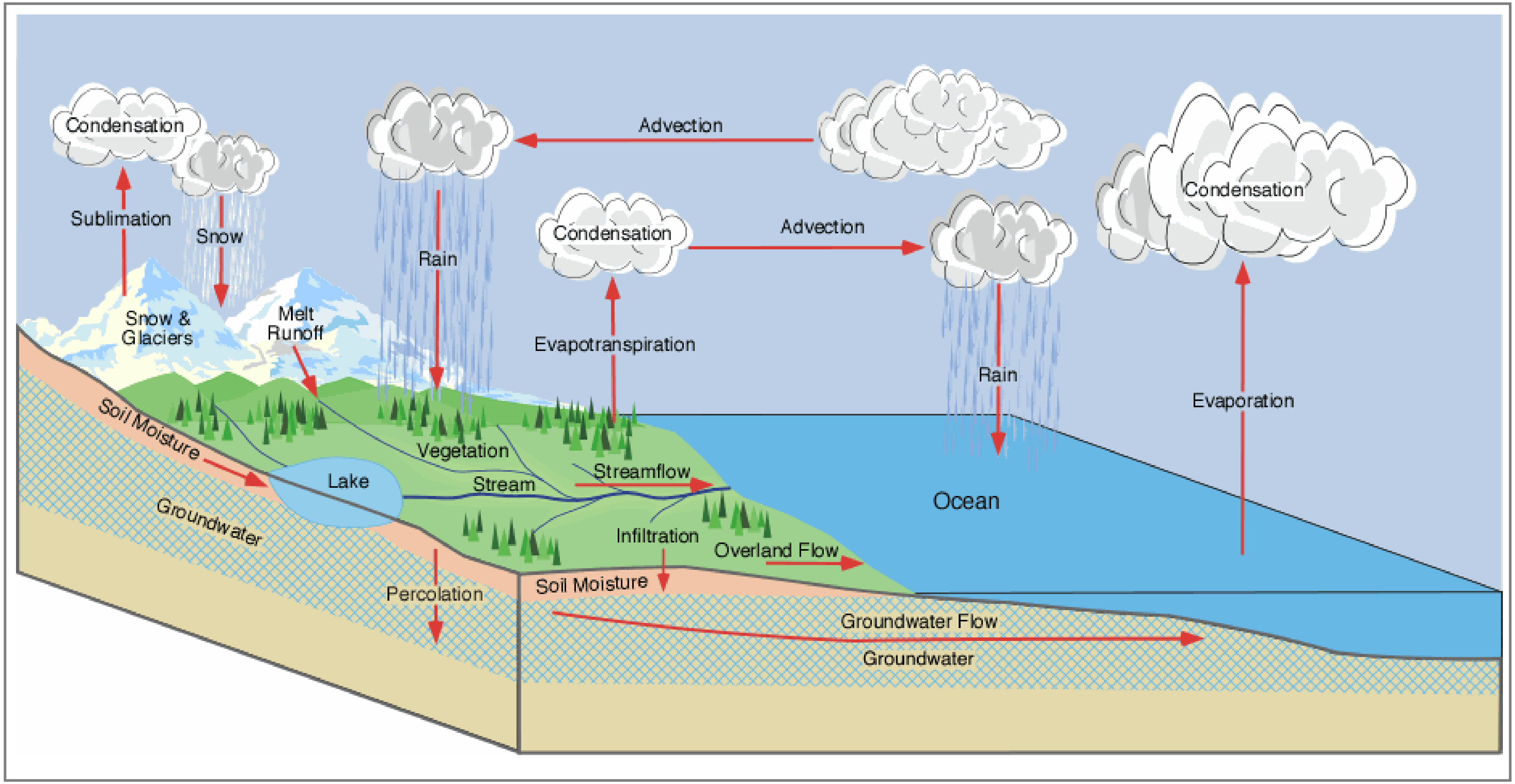


Figure 5-4 Schematic representation of a conceptual model of the hydrological cycle (Pidwirny, 2006).

5.5.1 The hydrosphere as a system

The hydrologic cycle is an example of a closed system with feedback (Figure 5-4). Water evaporates over surface water bodies of which the oceans form the biggest part. This water is carried inland via the atmosphere where it forms precipitation as rain and snow. The precipitation infiltrates into the soil and beyond to the groundwater zone, while the resultant produces surface water runoff or evaporate back into the atmosphere after being in storage in the soil.

Surface water infiltrates along the pathway to become soil water and groundwater while groundwater can discharge to the surface or form stream base flow or springs (i.e. feedback loops). The surface water and groundwater flows back to the sea where the cycle is closed (Figure 5-4).

Groundwater occurs as three main types. The first is the most important one, known as meteoric water that originates from the atmosphere as recharge from rainfall. The second is connate water, which were trapped during deposition and the third is juvenile water that originates from chemical reactions during volcanic eruptions (Zumberge and Nelson, 1976).

The hydrological cycle interacts between components and can be adversely affected if small changes are made in some of the components or pathways. It was found that evaporation and evapotranspiration could play a dominant role in reducing the groundwater component of base flow (AGES, 2010b). Evapotranspiration could be altered significantly if e.g. alien vegetation is introduced into the system. This indicates the importance of a systems approach as usually, groundwater considerations and models did not take evapotranspiration into account.

5.5.2 Why modelling? The importance of purpose

The purpose of using a model should be to turn data into information for decision-making purposes (Information Box 5-A). As shown earlier, the decision-making process requires data, expressed in terms of information until it is sufficient to make a decision (Section 3.6). Models are used to elevate the level of information that can be extracted from the data (Figure 5-5).

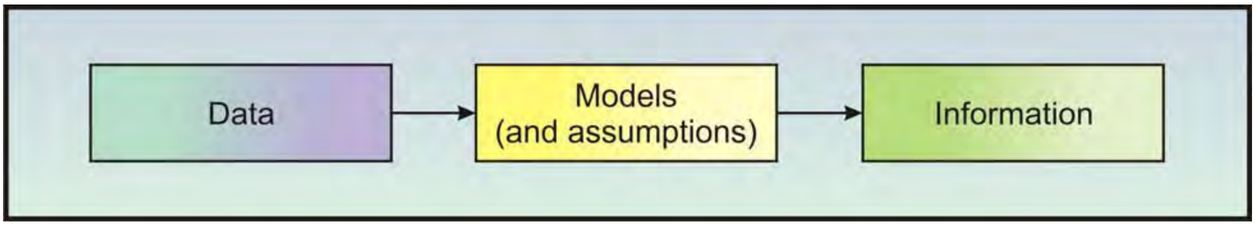


Figure 5-5 Schematic representation of the use of models to turn data into information.

One of the most common comments on modelling, is that there are idealised underlying assumptions and that it may not represent the physical system accurately. As indicated earlier, (Section 3.7) the role of assumptions is to substitute information, without which no model would be possible. It would only be a *perfect model* (which does not exist) that would not be based on assumptions. The purpose of the *application* of a model is to simulate the *problem*. The purpose is not to model the physical system with zero defect (Sterman, 2002). The purpose of *research* is to develop models that describe the physical system (i.e. porous and fracture flow models in groundwater) with ever increasing accuracy. It must be accepted that there is no model that will ever be able to simulate the physical system with exact precision. Modelling for the purposes of decision-making is therefore not a purely scientific exercise, but a management action that makes use of scientific tools to arrive at decision outputs.

To illustrate this point, a model can be equated to a map. A map is a model that represents an area in space. The purpose of a map is for the user to follow it to arrive at an unknown location. A simple line map that indicates the route/s between two points 1 and 2 is an example of a model (Figure 5-6). The map is a simplified, two-dimensional representation of a three-dimensional terrain (i.e. it is a model) that does not represent the physical area accurately with all the trees and traffic lights and cars etc. It can be stated by anyone that the map is *wrong*. The map is not *wrong* because the purpose of the map is to solve the problem of finding one of the two locations and not to represent the physical area exactly. A detailed map that is more accurate would be scientifically more correct or acceptable, as it represents the physical area more accurately (Figure 5-7).

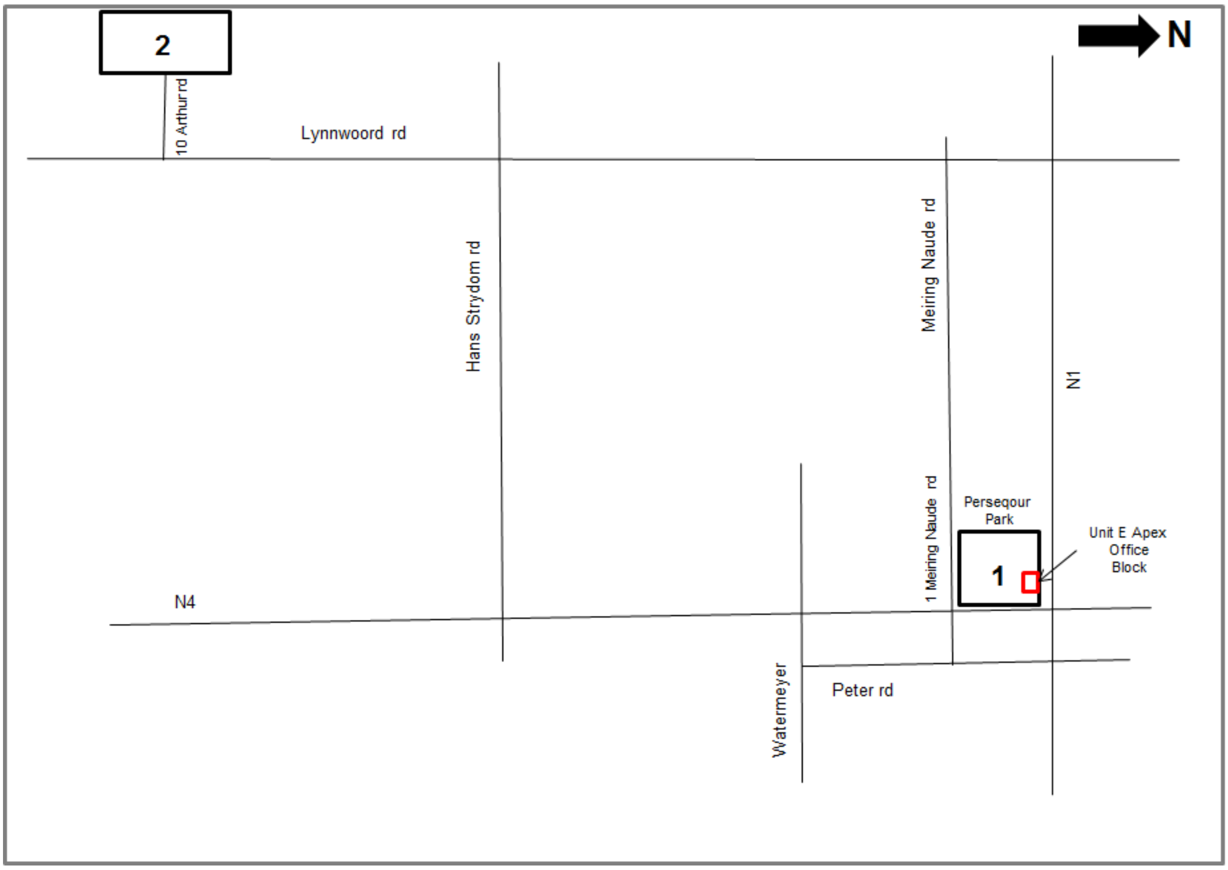


Figure 5-6 Schematic representation of a simple line map analogy for a model.

For the purposes of decision-making, the more detailed map could be an overkill as it would take more time (and cost) to compile while serving the same purpose. If e.g. the detailed map confuses the user because of too much data but not sufficient information, it would be a worse model than the simple line map. This is known as the *less-is-more effect*. The basis of this effect is that more information than is required could obstruct decision-making[40] (Goldratt, 1994; Taleb, 2010; Katsikopoulos, 2010; Beaman, et al. 2010 ; Section 3.3). It is not true that a model which describes the physical system most accurately is better, it is the one that is able to simulate the problem and provide the best answers for the purposes of decision-making, which is the better one. The complexity of the model is therefore not necessarily related to the complexity of the problem.

40 In experiments where people were shown pictures of objects, it was revealed that if say 5 clips of the same picture is shown, it led to better recognition than e.g. 10 clips in the same time span (Taleb, 2010).

Information Box 5-A

To further illustrate the point of purpose, if one were to ask anyone whether a knife is a dangerous object or not? The answer could be that people get injured or even murdered by using knives and therefore all knives should be banned everywhere. If say an innocent person gets mugged and stabbed with a knife in the street. An ambulance takes that person to the trauma unit. The surgeon arrives and what does he use to open and cure the wound? A surgical "knife". It is therefore the purpose of the object that determines whether it is good or bad and not the object itself. It is the same with modelling. Everything has a use that can be abused if applied outside its purpose.

Figure 5-7 Schematic representation of a detailed map analogy of a model.

5.6 The use of systems models in decision-making

As discussed earlier, decision-making in environmental management is not a scientific exercise. It is a combination of the management of the effects that humans have on the environment, supported by scientific tools and facts (Section 2.10). The first step in systems analysis is to develop an interaction map[41] for the specific problem. In groundwater analysis, this is known as conceptual model development (Section 2.9.8.1, Figure 2-14, Figure 2-15). A process flow diagram (PFD) of the modelling process shows that (Figure 5-8):

1. A proper definition of the problem is the first step in the modelling process. The model will be incorrect if the problem definition is not done correctly. Part of the problem definition, is to determine the problem boundaries. If a problem cannot be bounded, it cannot be solved.
2. Following the definition of the problem boundaries, the model purpose must be defined. A model without a decision-making purpose has no practical use.
3. An evaluation of the physical system is important to determine which model should be used. A saturated groundwater flow model cannot be used for an unsaturated problem and vice versa.
4. Once the physical system is evaluated, then existing data should be analysed before new data is collected. If there are no existing data, then a sub-minimum data should be collected for a first iteration of analysis. It is important that data analyses should be maximised by using appropriate statistical methods.
5. Based on the physical data, develop a conceptual model.
6. Based on the conceptual model, develop or select a mathematical model.
7. Use the mathematical model to construct a problem-specific model.
8. Calibrate the model using the data that was collected.
9. Develop simulation or management scenarios.
10. Simulate scenarios and evaluate results.

[41] Also known as a strategy map.

11. Evaluate sensitivity and importance of input parameters and data. Perform data worth exercise.
12. Iterate back to Step 4 to collect and analyse data if required. It is during this step only that data collection becomes important for now it can be optimized of sufficed.
13. Re-iterate between steps 8 to 11 until sufficient or optimal and modelling process in terms of data has converged.
14. Provide and evaluate results.
15. Make decision or re-iterate to step 1 and step through the process until the decision-making process converges.

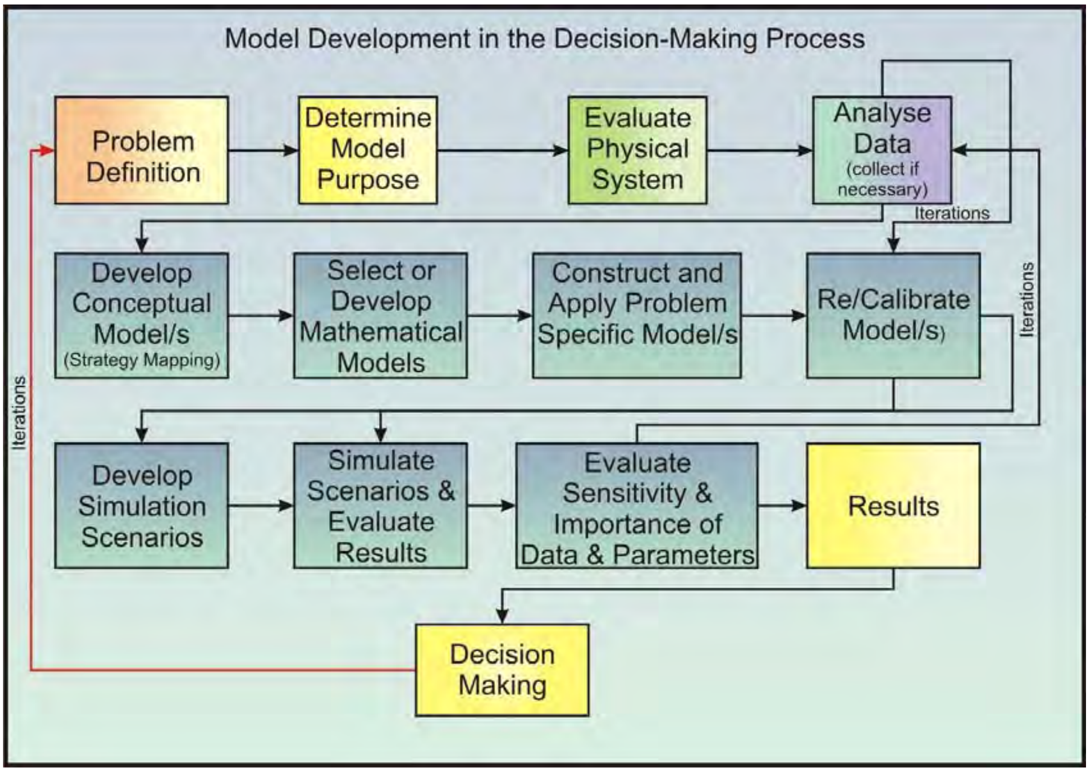


Figure 5-8 Schematic representation of systems modelling in the decision-making process.

Due to the nature of the physical data, the modelling process for decision-making will have the same logarithmic characteristics as detailed in the previous section (Section 3.6). The process follows a Bayesian approach (Section 2.9.4) and should be stopped once sufficient or optimal output has been reached for the purposes of decision-making (Section 3.6, Figure 3-26).

This process will be described in more detail in the next chapter with the use of a practical problem.

5.7 Discussion

Systems thinking can be applied on decision-making in complex environmental problems as it is used to understand complex systems. In a classical linear cause-effect view, secondary problems are generated while aiming to solve a primary problem. A systems approach allows the consideration of cyclic processes where feedback plays an important role. This assists in the identification of secondary or downstream effects.

Natural and environmental processes can be described by systems theory. System models are valuable scientific tools that can be used to simulate environmental problems. It makes provision for the consideration of sub-systems which can be represented by e.g. the groundwater component within the hydrological cycle. The purpose of a model is important, as it guides the modelling process. Models are used to simulate the problem and not necessarily the physical system. Simple models that focus on the problem can often provide more information than complex models that focus on the physical system. This is known as the less-is-more effect. Models are useful tools in the decision-making process that assist the analyst to see beyond the horizon, but must be applied within a decision-making framework.

CHAPTER 6

"If there is more than one solution to a problem, the simplest one is usually the correct one" Unknown

6 APPLICATION OF A SYSTEMS APPROACH ON ENVIRONMENTAL DECISION-MAKING AT THE FAR EAST RAND BASIN (FERB) MINE WATER FLOODING PROBLEM

6.1 Introduction

Decision-making in complex environmental problems is a conundrum that faces most environmental managers. The decision-making process characterised in previous chapters indicated that it is not possible to obtain perfect information or to pursue the perfect decision. It was determined that the decision-making process follows a logarithmic trend that converges to perfect information that it never reaches. The use of systems thinking and systems models is evaluated for the purposes of environmental decision-making. Important aspects that are evaluated in this section are to determine how decisions can be made on complex environmental problems with limited data by using a systems approach.

The use of scientific methods and models is tested as a decision-making tool based on the Far East Rand Basin (FERB) mine flooding and decanting problem. An assured methodology is developed that can be used to make decisions with limited information.

6.2 Application of a systems model on the FERB flooding and decanting problem

6.2.1 General

The FERB flooding problem was identified in the 1990's as one of the largest environmental water problems that will be encountered in South Africa in the future. Estimates of the time scale when the problem would occur was vague as the mines were still operational. In 2004 to 2006, a series of investigations were undertaken by the Department of Mineral Resources (DMR) to characterise the problem. The scientific studies provided information that was contradictory. The result was that no long-term decisions or action were taken by the political decision-makers with regard to management and mitigation measures. The problem has reached the stage where the lack of decision-making resulted in media reports that created hype and a fog of misunderstanding. The result is government intervention without a proper understanding of the problem. In this section, a systems approach is followed within the decision-making process defined in Section 5.6 to illustrate the usefulness in decision-making.

6.2.2 Problem and boundary definition

The project was initiated to determine what the origin of water inflow into the FERB in Johannesburg is and to identify what could be done about it. The ingress of water into the mine void creates a large water management problem as the water that flows in reacts with the rock mass in the old mined-out stopes that produces Acid Mine Drainage (AMD) with contaminated water that is pumped out as part of the mine dewatering process (Figure 6-1, Figure 6-2, Figure 6-3).

The objectives of the investigation were to (AGES, 2005b):

1. Delineate the problem boundaries and identify the study area limits.
2. Determine the water ingress areas and potential sources of ingress.
3. Distinguish between surface water and groundwater ingress or recharge.
4. Flooding rates and times.
5. Decanting locations and expected outflow rates.
6. Identify possible sustainable solutions to the problem.

The boundaries of the problem are located within the surface water catchment boundaries of the Blesbokspruit, which is located in the Vaal River Primary Catchment (C) (Figure 6-4, Figure 6-5). The problem and area boundaries cover a total area of 1400 km^2 that includes quaternary catchments C21D (445 km^2), C21E (628 km^2), C22C (196 km^2) and C21F (123 km^2). Topographic elevated surface watersheds are formed towards the east, west and north and were included as no-flow boundaries in the assessment. Outflow boundaries were considered along the surface water drainages to the south via the Blesbok Spruit and the Withok Spruit.

6.2.3 Determine model purpose

The purpose of the model was to determine management and mitigation measures for the regional environmental water problem. The aim was to determine where and when the decanting would occur and what could be done to manage and mitigate the problem. It was understood that the physical system is highly complex with three-dimensional, layered aquifer systems and underground mining developments that consists of shafts, tunnels and stopes. It is considered as a first iteration of problem quantification with the main aim to assist in the understanding of the problem and identification of sustainable management and mitigation measures.

6.2.4 Evaluate physical system

The physical system consists of (Figure 6-1):

- Water inflow that is controlled by atmospheric precipitation and evaporation processes.
- Underground mines with open mine voids, shafts and tunnels.
- Surface water streams, wetlands and dams.
- Groundwater systems consisting of aquifers, aquitards and preferential pathways formed by cracks and fissures.
- Soil and unsaturated zones that form a barrier to seepage and infiltration.

The physical system requires an approach that takes account of precipitation, evaporation, surface water flow, groundwater flow and flow through underground voids. The groundwater component in a systems approach can therefore not be considered in isolation.

6.2.5 Analyse data (and collect if necessary)

Collect data on the important components identified in the previous point. The data collection should include surface water and groundwater components with pathways. In the field study, data that were collected included (Table 6-1):

- Surface water features, flow rates and water quality.
- Mine dewatering rates with time.
- Borehole co-ordinates, water levels and water quality.
- Aquifer tests and groundwater parameters.

Table 6-1 Data checklist used in the assessment[42].

No	Model data input	Source and assumptions	Date
1	Catchment area	WRMS 2000 quaternary catchment areas	2006
2	Rainfall	Data received for all rainfall stations in the study area from SA Weather Bureau	2006
3	Recharge	Estimated from minimum recharge rate values for Dolomitic and Karoo aquifers based on Bredenkamp et al. (1995)	1995
4	Dams	Digitised from Google Earth	2006
5	Geology (primary) area (km^2)	1:250 000 Geological map. Shape files obtained from Pamodzi Gold geological unit	2006
6	Depth to water level (m)	Determined from 174 boreholes obtained from the Council for GeoScience and 23 newly drilled boreholes. The average depth to water level is 17.5 m with a minimum of 0.8 m and a maximum of 150 m (AGES, 2006)	2006
7	Min depth to water level (m)	Determined at 0.8 m from surface based on hydrocensus	2006
8	Max aquifer depth (m)	3500 m, which is below the maximum mining depth (AGES, 2006)	2006
9	Water level management constraint (m)	The post-operational water level management constraint was set at the BCL which is at 1500 mamsl.	2006
10	Aquifer Transmissivity m^2/d	Determined from pumping tests at 11 boreholes (AGES, 2006)	2006
11	Aquifer storativity (1)	Assumed to be 0.001 as an average for the Karoo Aquifers and 0.01 to 0.008 for the dolomitic aquifers (Bredenkamp et al. 1995). The mine stopes were set at a storativity of 1 as it represents and open volume.	2006
12	Groundwater volume in storage (m^3)	Calculated from the aquifer area and storativity values	2006
13	Max usable groundwater volume in storage (m^3)	Not applicable	2006
14	MAP (mm/a)	Calculated from rainfall station data obtained from the SA Weather Bureau	2006
15	MAE (mm/a)	Calculated from rainfall station data obtained from the SA Weather Bureau	2006
16	MAR (%)	WRMS 2000 quaternary catchment areas	2010
17	MAR (m^3/a)	Calculated	2010
18	MAR (mm/a)	Calculated	2010
19	Rainfall 95% assurance (mm/a)	Not applicable	2010
20	Recharge 95% (m^3/a)	Not applicable	
21	Number of dams	Delineated from the 1:50 000 topographic maps	2006
22	Avg seepage (mm/a)	Determined from double-ring infiltrometer tests	2006
23	Dam Seepage Area (km^2)	Digitised from Goolgle Earth and Geographic Information(GIS) maps for all the existing dams.	2006
24	General autho- rizations	Not applicable	
25	Number of abstraction boreholes (Other)	Not applicable	2006
26	Total borehole abstraction (m^3/a)	Not applicable	2006

[42] The components that were marked *not applicable*, were kept in as part of the checklist function but were not applicable for the purposes of this study.

No	Model data input	Source and assumptions	Date
27	Number of livestock farms	Not applicable	2006
28	Total livestock farm usage (m^3/a)	Not applicable	2006
29	Number of mines	Council for GeoScience data base	2006
30	No of communities	Not applicable	
31	People in community	Not applicable	
32	Total community borehole usage (m^3/d)	Not applicable	
33	Average Farm irrigation area (ha)	Not applicable	
34	Average Alien Vegetation (ha)	Not applicable	
35	Average Forestry area (ha)	Not applicable	
36	Average Riparian Alien veg (ha)	Not applicable	
37	Wetlands (Ground water) (km^2)	Delineated by Wetland Consulting (2005)	2008
38	No of springs	Not applicable. Most of the springs are dry	2006
39	Spring flow (m3/a)	Zero	2006
40	Evapotranspiration flow loss (m^3/a) % of catchment	Calculated from the MAE and surface area of water bodies	2006
41	Net GW base flow (m^3/a)	Simulated with the numerical model	2006
42	Net Base Flow Required by EWR - Drought low flows (m^3/a)	Not applicable	

The water level data was interpolated across the model domain using kriging. Water levels and transmissivity values were interpreted statistically by determining averages and standard deviations (AGES, 2006).

6.2.6 Strategy mapping, process mapping or conceptual model development

One of the important steps in the modelling process is what is known as process flow development in engineering or strategy mapping in business or conceptual modelling in geohydrology (Section 2.9.8.1). This process assists and trains the analyst to map out the physical components that influence the system, as well as the interactions and processes. In this first step in the modelling process, a process flow diagram (PFD) is developed that assists in the understanding of the problem (Figure 6-1).

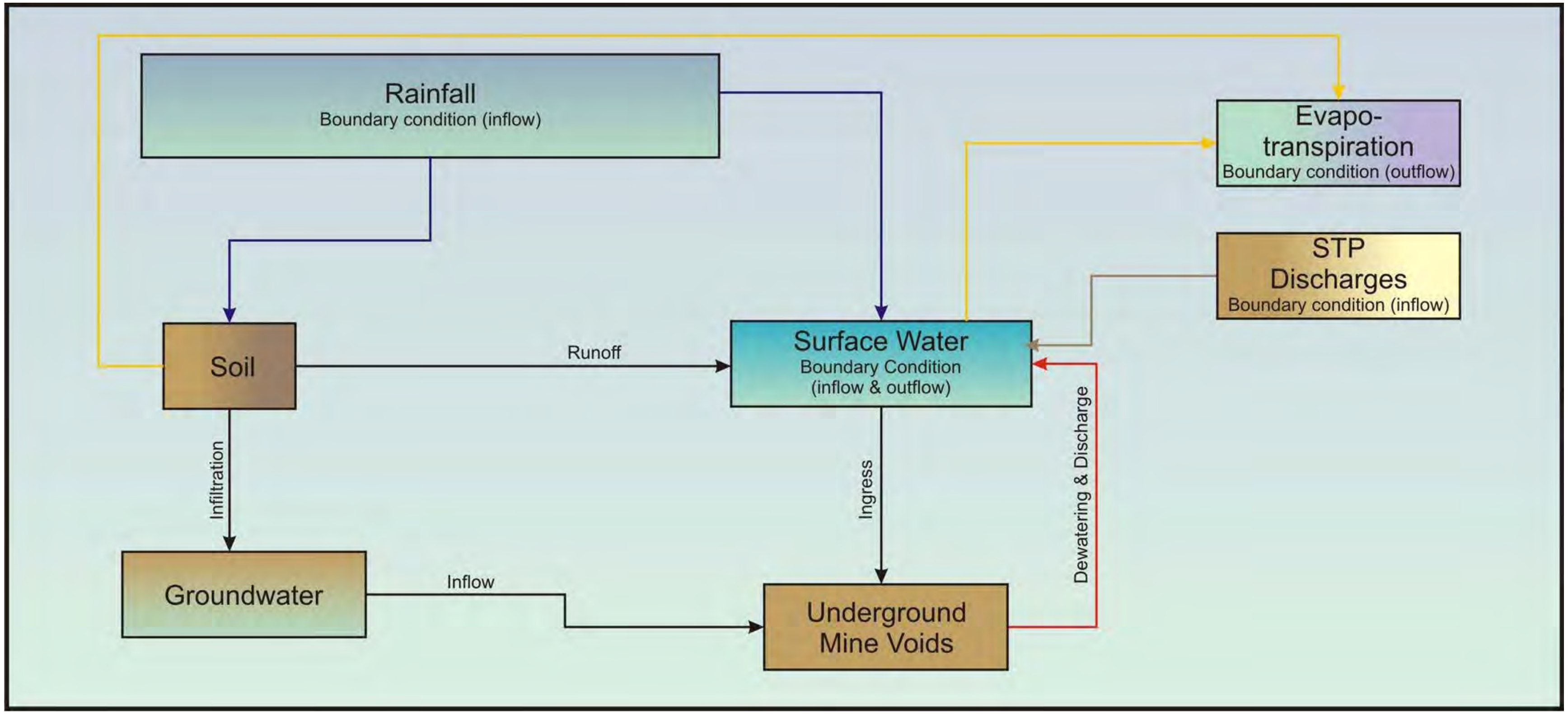


Figure 6-1 Schematic representation of a conceptual process flow model.

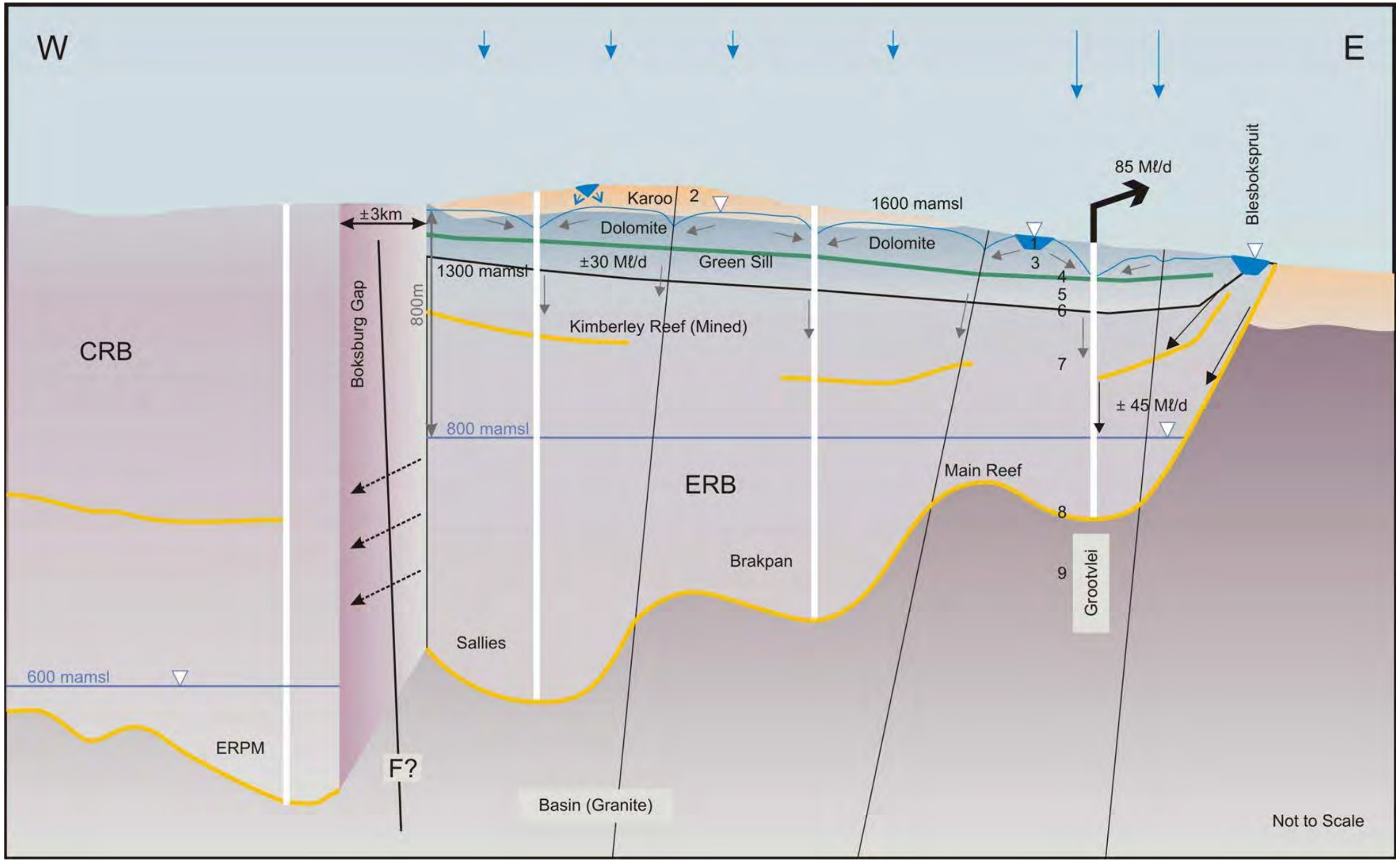


Figure 6-2 Schematic representation of a graphical conceptual model representing the system (AGES, 2006).

	A	B	C	D	E	F	G	H	I	J
1	West Complex Tailings Dams		C.1					H.1		
2		Western Extension Tailings Dam	C.2					H.2		
3			Bokkamp Dam			F.3		H.3		
4				Waste Rock Dump		F.4		H.4		
5					Skelm Dam	F.5		I.5		
6	A.6	B.6	C.6		E.6	Surface water	G.6		I.6	
7						F.7	Soil	H.8		
8							G.8	Groundwater	I.9	
9									Vaal River	
10										Schoonspruit
	Legend									
	Weight	**1**	**2**	**3**	**4**	**5**	**Source**	**Pathway**	**Receptor**	

Figure 6-3 Interaction matrix conceptual model for environmental geohydrological interaction (AGES, 2006).

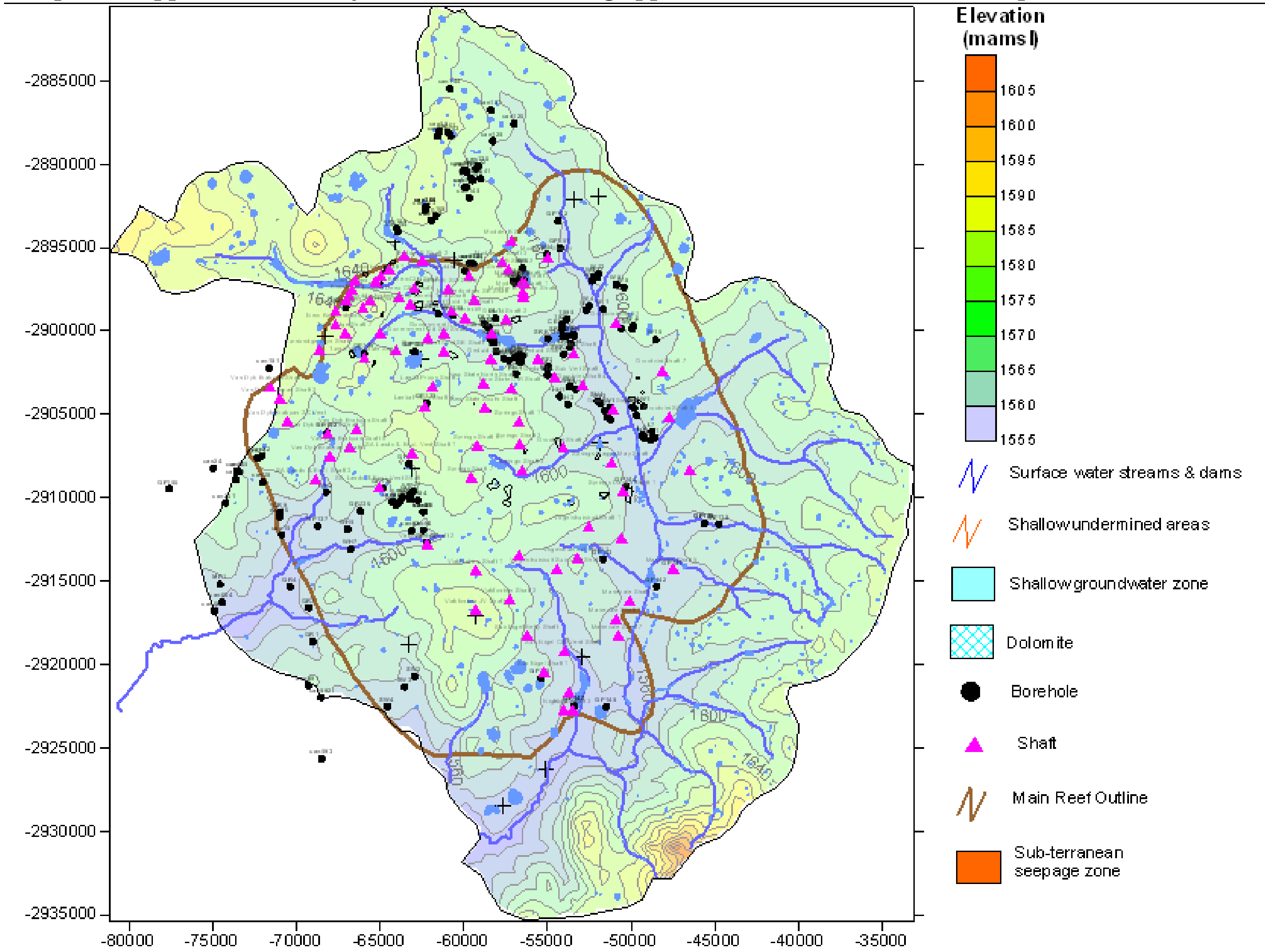


Figure 6-4 Far East Rand Basin study area and surface catchment with topography (AGES, 2006).

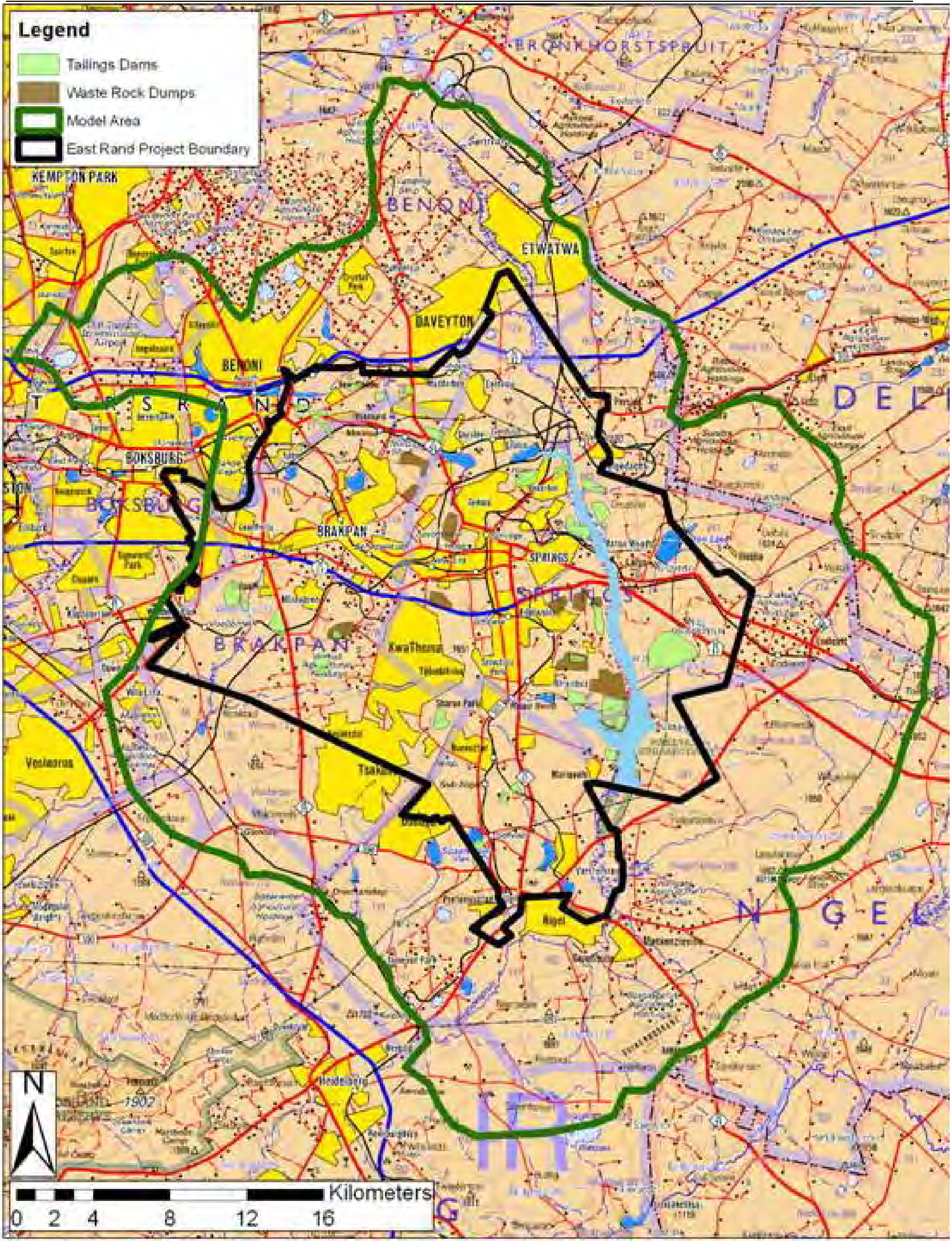


Figure 6-5 Topographic map of the Far East Rand Basin catchment (AGES, 2006).

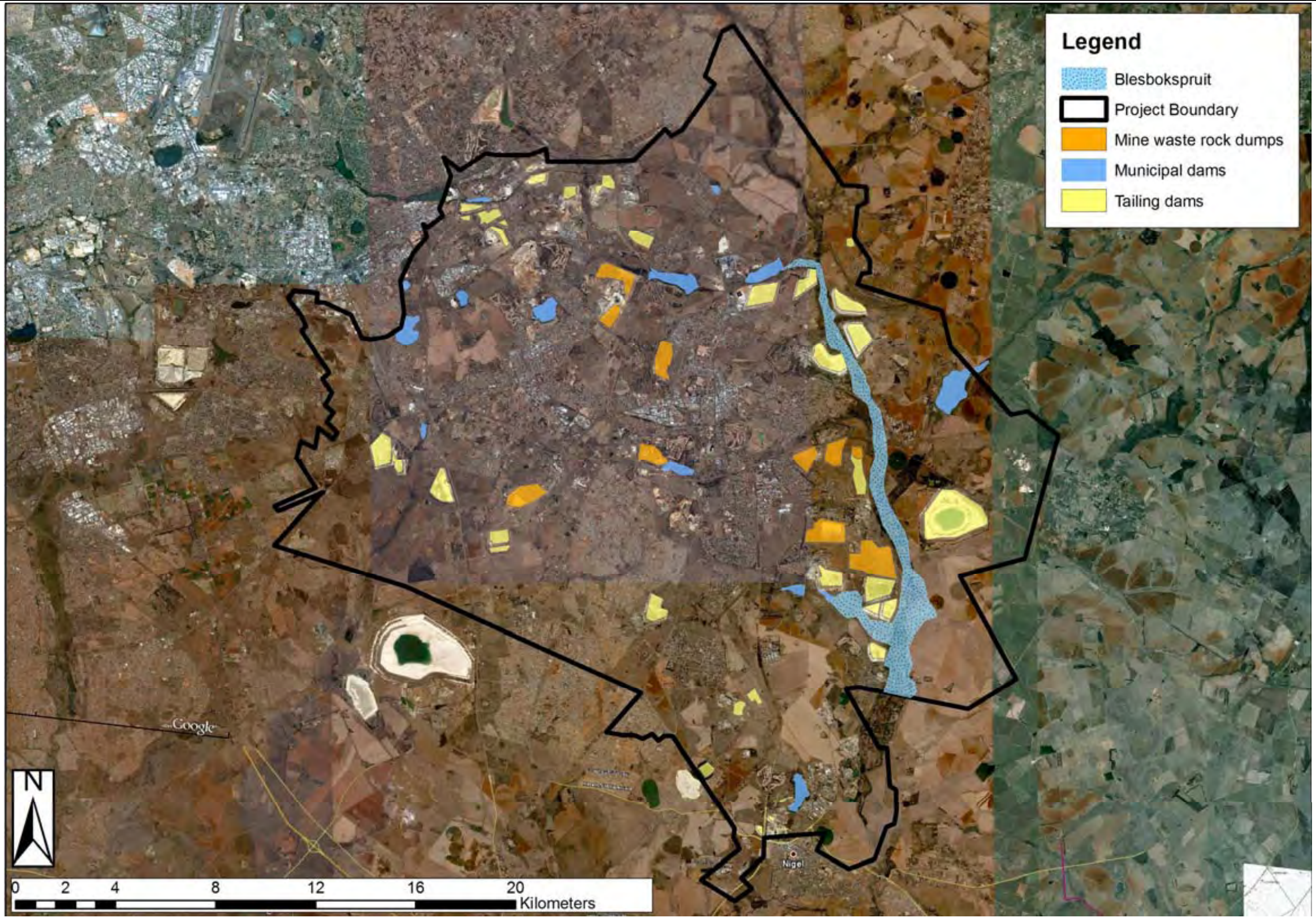


Figure 6-6 Google Earth (Pro) image of the Far East Rand Basin mining areas.

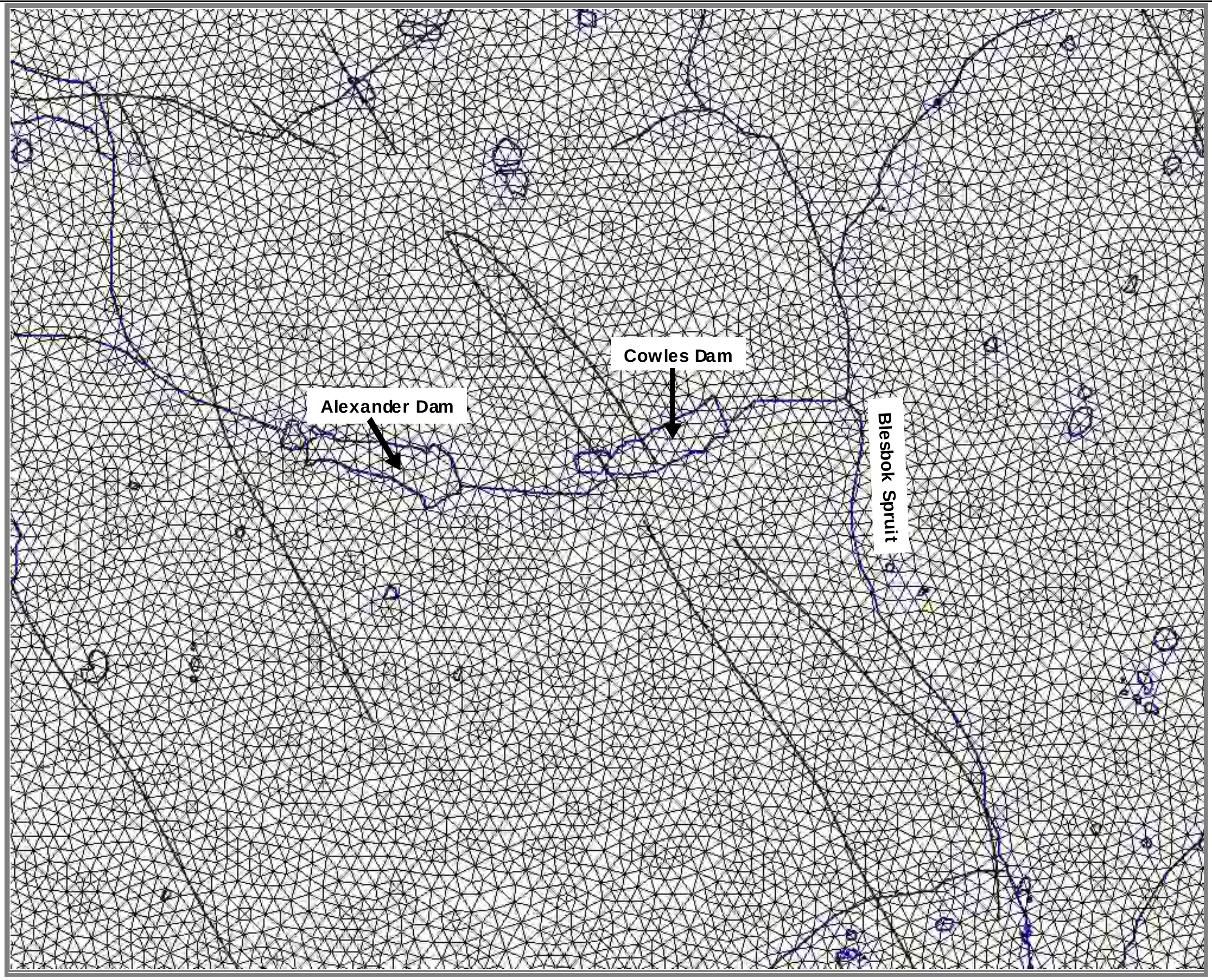


Figure 6-7 Zoomed view of the finite element network in the Grootvlei Mine area, showing the resolution (AGES, 2006).

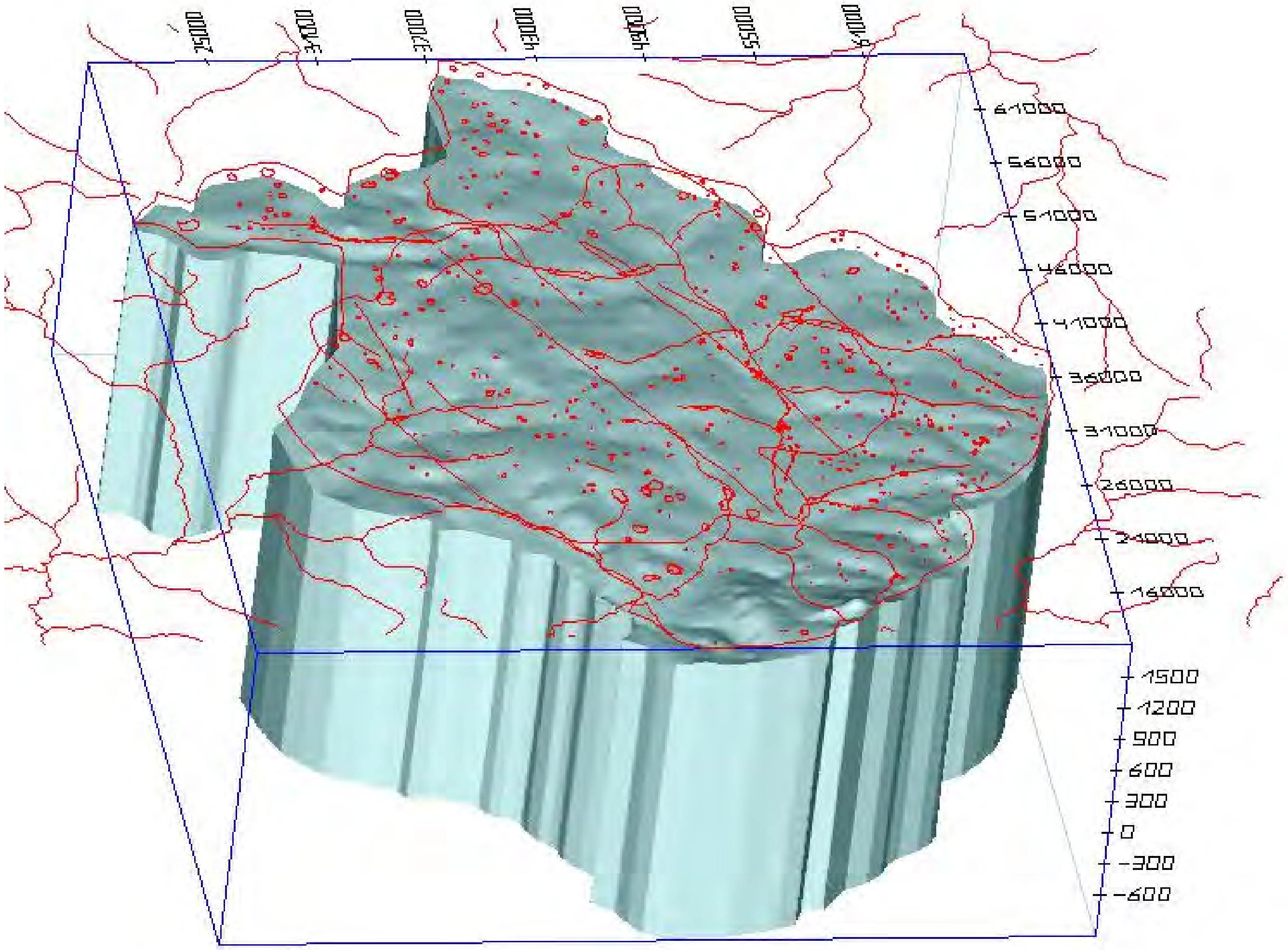


Figure 6-8 FERB three-dimensional spatial model (North is at the top of the page) (AGES, 2006).

During the development of the process flow diagram, the boundaries (i.e. sources), pathways and interaction directions and components must be identified. The process flow diagram or interaction matrix model is then extended to compile a site specific graphic representation of the geometry of the system (Figure 6-2; Figure 6-3). In the conceptual model, the boundary conditions and processes are identified that form the basis for the development of the mathematical model.

6.2.7 Select or develop mathematical model

Based on the conceptual models, appropriate mathematical models are selected. The models that are selected would be based on surface water, groundwater or a combination. In this case, the surface water was modelled based on an analytical, steady-state water balance. Due to the importance and complexity of the groundwater, it was simulated three-dimensionally with the surface water components acting as boundary conditions. In a systems model, specific components can be evaluated in more detail with feedback back into the main model. It is however important not to consider one component, such as groundwater in detail without consideration of the other important components.

6.2.7.1 Groundwater component

The mathematical model used in the groundwater sub-component is implemented in a numerical (finite difference or finite element) simulation system. In this case, the FeFlow (Version 5.2; Diersch, 2002) code was used for the simulations. It simulates groundwater flow, contaminant mass and heat transport processes as coupled or separate entities. It is based on the physical principles of the conservation for mass, chemical species, linear momentum and energy in a transient (time dependent) and spatially in a three-dimensional numerical analysis. Groundwater flow in an inhomogeneous anisotropic confined aquifer is mathematically expressed as (Bear, 1979; http://www.humboldt.edu/geology/courses/geology556):

$$\frac{\partial}{\partial x}\left(K_x \frac{\partial h}{\partial x}\right)+\frac{\partial}{\partial y}\left(K_y \frac{\partial h}{\partial y}\right)+\frac{\partial}{\partial z}\left(K_z \frac{\partial h}{\partial z}\right)=Ss\frac{\partial h}{\partial t} \qquad (5\text{-}1)$$

with the following notations:

h	= hydraulic head	[L]
K	=hydraulic conductivity	[L/t]

S_0	= specific storage coefficient	[L^{-1}]
b	= Aquifer thickness	[L]
t	= time	[T]
x,y,z	= spatial dimensions	[L]

6.2.8 Construct and apply problem specific model

The mathematical numerical model is defined spatially based on the geometry of the physical system as determined during the definition of boundary conditions and conceptual model development Step 5. During this step, the conceptual model is quantified with the required input and output parameters. The study area was defined as the surface water catchment, which is situated sufficiently outside of the underground mined out areas (Figure 6-4, Figure 6-7). An assumption was made that no groundwater inflow from outside the surface water catchment occurs. This assumption could be questioned, but due to the complexity of the problem, it could not be solved without this assumption (Section 3.7). This assumption is scrutinised in Section 6.3.2. The study area is then differentiated into the smaller spatial components sufficient to model the problem (Figure 6-7). It is important to note that the required resolution to solve the problem must be pre-defined in terms of the purpose of the model.

Based on the conceptual model, a three-dimensional finite element model was developed that consisted of 9 layers (Figure 6-8).

6.2.9 Calibrate or recalibrate model

The model calibration process is done by changing input parameters and or boundary conditions so that the behaviour of the model would be representative of the actual system. This is done by fitting steady-state and or transient model parameters such as water levels or water quality with time against measured data. The changing of model parameters until an acceptable fit is obtained between the simulated and measured data (Figure 6-9). The calibration process follows the same pattern as the decision-making process, as it converges with time (Section 3.6, Figure 3-25). It also adheres to the law of diminishing returns (Section 3.3.5). As the calibration process continues, it should converge to a point where no further improvement of calibration is practically possible. Beyond that point, the calibration process will deviate from the decision-making process and it would start to diverge if more changes to parameters or boundary conditions are made. A calibrated model contains uncertainty in that there are usually a large number of combinations of parameters that would produce a good fit. This is where it is important to pin down uncertain parameters with measured data, usually from field tests.

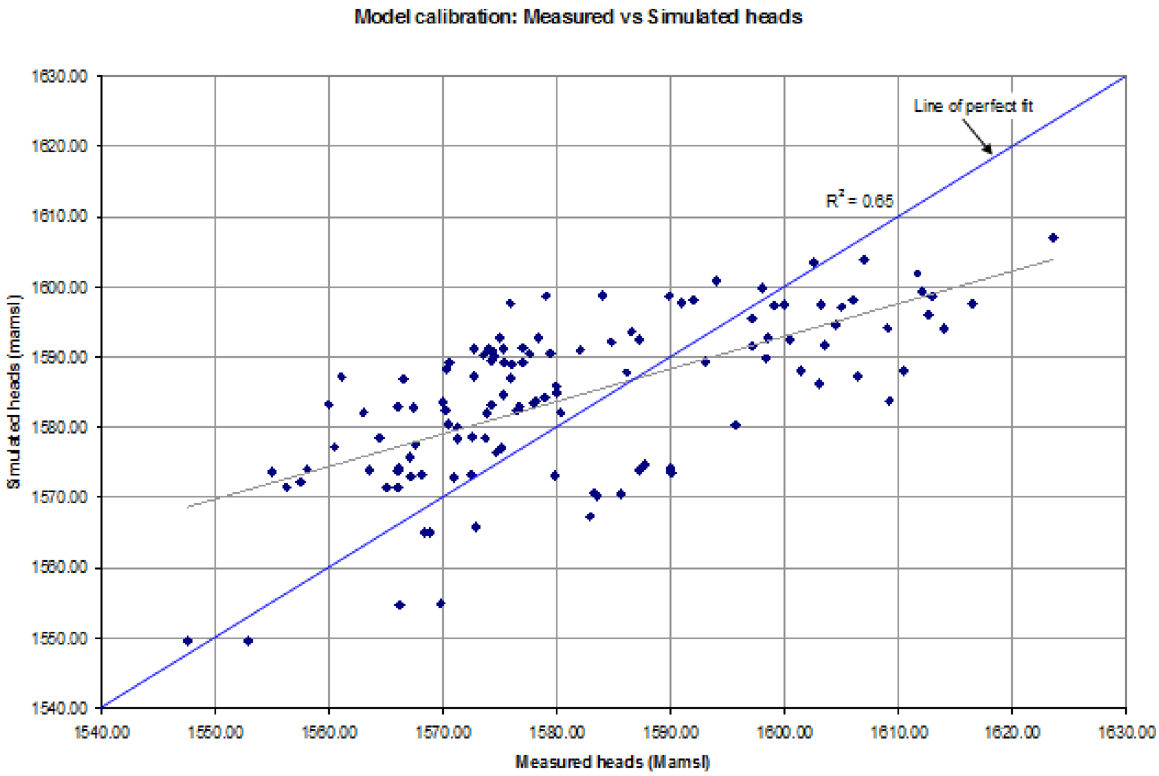


Figure 6-9 Far East Rand Basin steady-state simulation: Measured vs simulated heads (AGES, 2006).

6.2.10 Scenario development and testing

This section is one of the most important steps and it will be evaluated in more detail than the other steps in this section.

Formal scenario development was first used in the Second World War for war game analysis (Van der Heijden, 1996; http://www.unep.org/ieacp/iea/). Scenario planning was further developed by Shell in the 1970's. The purpose was to develop plans to identify and manage potential future risks. At the time, the oil price was very high and Shell used scenario planning to determine strategies to cope with the possibility of OPEC reducing oil supply and raising prices. This was an event that none of the other oil companies thought of. When this eventually happened in 1973, Shell grew from the eighth largest oil company to second in just two years. The benefits of scenario planning in business decision-making (Section 2.6) was obvious to the world and since then, it is widely used (Drinkwater, 2003).

Once a model is calibrated, it can be used for scenario[43] (i.e. what if) testing. In the modelling process, it is accepted that it is not possible to simulate the future behaviour of the real system, especially when prospective[44] evaluations are made. All we know about the future is that it will be not the same as today and it will be unexpected. Scenarios used in models *are not used to predict the future*. We have to accept that future prediction is impossible (Drinkwater, 2003; Taleb, 2010). It is more sensible to develop futuristic scenarios that can be tested using models and the aim of scenarios is not to predict. In most cases, two or more contrasting scenarios are developed to highlight risks and uncertainties. In a groundwater supply scenario, lower transmissivity and recharge values will be used where in another scenario for the same model, higher transmissivity values and flood rainfall recharge conditions might be considered to evaluate e.g. the potential for mine dewatering volumes. Scenarios in the business environment are defined as;

"Scenarios are tools for ordering one's perceptions about alternative future environments in which today's decisions might be played out… Scenarios resemble a set of stories, written or spoken, built around carefully constructed plots… Good scenarios are plausible and surprising, they have the power to break old stereotypes, and their creators assume ownership and put them to work. Using scenarios is rehearsing the future. By recognizing the warning signs and the drama that is unfolding, one can avoid surprises, adapt and act effectively" Global Business Network (www.gbn.org/public/gbnstory/scenarios)

Management scenarios are developed that could be simulated and tested to evaluate the system response. The purpose of management scenarios is to determine what could be done to manage the system if future inputs, outputs or internal parameters change. In the case of the FERB water problem, it would be important to determine what would happen if abstraction of contaminated mine water at depth is stopped.

As experienced by Shell in the 1970's, the process of scenario development is important as it could provide answers that are critical to the decision-making process. Scenario development should be based on a systematic approach to ensure that outcomes can be evaluated and the

[43] A scenario is a description of how things might happen in the future (Oxford English Dictionary, 2006).
[44] Note that the word prediction is avoided on purpose as it is accepted that models cannot predict. It can be used to evaluate prospective scenarios (Dr M Kozak, personal communication).

process audited in the decision-making process. Scenario development should therefore be used as a strategic planning tool (Section 2.3.1).

In radioactive waste disposal, a formal methodology for scenario development exists that is based on features, events and process (FEPs) (Van Blerk, 2000). A feature is e.g. a surface water body, an event would be represented by e.g. rainfall and a process would be represented by e.g. groundwater flow. The potential effects of any changes on FEPs e.g. a dam break or a flood can be evaluated using the simulations. The significance of the outcome is then used for decision-making.

Systems models can be used to simulate scenarios with non-tangible effects such as socio-economic and political influences of various stakeholders (Szczerbicki, 2004; Levine et al. 2008; Elias, 2008).

6.2.10.1 Scenario development

The development of scenarios should be structured so that the output can be measured against a starting point known as a *base or reference case*. The following steps are required in scenario development (adapted from Drinkwater, 2003) and based on the actual case study:

1. Determine purpose of scenarios: The purpose of the scenarios in the field investigation is to evaluate what the possible sources of the underground mine waters were.
2. Setting the scene for scenario development: The scene is represented by a highly disturbed land use, surface water and underground system if compared to natural conditions. It is important to determine a base case scenario that can be used as a reference point.
3. Identify the key driving forces: The key driving forces are represented by natural surface water, artificial discharges, groundwater and water quality. The process flow diagram and conceptual model development assists in the identification of these driving forces (Section 6.2.6, Figure 6-1).
4. Isolate the driving forces and rank by importance and uncertainty: The most important driving forces is the interaction between surface water and groundwater.
5. Develop or select scenarios: The following scenarios were developed:

i. Scenario 1: Pre-operational surface water and groundwater flow to evaluate the dynamics of the system prior to mine developments.

ii. Scenario 2: Present day situation with surface water and groundwater flow to evaluate the effects on mine dewatering. This was also the base case scenario since this is the present case.

iii. Scenario 3: Future potential for mine rewatering and decanting to determine where water would decant, when and at what volumes.

iv. Scenario 4: A re-evaluation of Scenario 2 with the consideration of economic and social (legal) aspects. These aspects are in some cases less tangible but can have a more important influence on the management options or it can form key constraints.

6. Review and critically evaluate scenarios. The critical evaluation of the scenario results is done where decision-making methodologies used in business management (Section 2.6) and judicial processes are used (Section 2.7). This is done to test the validity of the scenario outcomes.

6.2.10.2 Scenario testing

In the following section, the overall water balance was modelled analytically, while the groundwater inputs (components listed as groundwater and underground voids, Figure 6-14) were obtained from a more detailed numerical groundwater flow model. The models were used to run the specific scenarios to evaluate the outcome/s. The water balance contains all inflow and outflow components and must balance to provide an evaluation of the status of the system.

The water balance approach is done to evaluate the regional water balance based on a systems approach where boundary conditions (sources and sinks), components, pathways and interactions between these were considered. The system water balances were based on annual volumes that were averaged to daily values (Table 6-4, Figure 6-13, Figure 6-14, Figure 6-16). This was done by using measured or calculated values.

6.2.11 Scenario 1: Pre-operational surface water and groundwater flow

6.2.11.1 Scenario 1: System water balance for pre-operational phase

The system water balance is driven by the boundary condition sources of rainfall, evaporation, downstream surface water and downstream groundwater. The system water balances were developed to determine the potential flows and interactions as follows: (Table 6-4, Figure 6-13, Figure 6-14, Figure 6-16).

Rainfall: A total of 2936.5 Mℓ/d on average is received from rainfall on the soil zone.

Soil: 2919.1 Mℓ/d falls on soil. From the soil zone, most (2685.1 Mℓ/d) of the water is evaporated while 146 Mℓ/d is received by surface water bodies as runoff. The resultant 88.1 Mℓ/d infiltrates to the groundwater zone as recharge.

Site surface water: A total of 17.4 Mℓ/d is received directly from rainfall and 146 Mℓ/d from soil surface runoff. A large component of 22.2 Mℓ/d evaporates and 17.6 Mℓ/d, is received from the groundwater component of base flow.

Groundwater: Recharge from rainfall of 88.1 Mℓ/d is received via soil infiltration beyond the root zone. The recharge is balanced by the groundwater component of base flow of 88.1 Mℓ/d and 70.5 Mℓ/d is lost to evapo-transpiration in the interaction zone. The site groundwater receives an estimated 1 Mℓ/d from the upstream groundwater.

Downstream surface water: The downstream surface water acts as a boundary condition (sink) to the system which receives 163.4 Mℓ/d from the site surface water.

Downstream groundwater: The downstream groundwater acts as a sink which receives approximately 11 Mℓ/d from groundwater exchange with the local but regional aquifer.

6.2.11.2 Scenario 1: Groundwater balance for the pre-operational phase

The detailed groundwater flow balance (groundwater and underground mine void) were evaluated as a sub-system that feed back into the main model (Figure 5-2). It provides more accurate flow figures and is differentiated into more flow components that are important for the mine void characterization.

The inflow components of the pre-operational groundwater flow balance are controlled by recharge from rainfall, which occurs on the Karoo and other Fractured Aquifers and the regional

Malmani Dolomite Aquifer (Figure 6-4, Figure 6-5). The simulated groundwater recharge totals 88.1 Mℓ/d (no's 1 to 4 in Table 6-2). The area covered by dolomite is 72 km^2 on the Blesbokspruit catchment and 40 km^2 for the Withokspruit catchment (Figure 6-4). The surface water inflow component is small at 5 Mℓ/d as the flow gradient is mainly from the aquifers to the streams. The outflow component in the natural groundwater system is balanced by the groundwater component of base flow and losses (88.1 Mℓ/d) to surface water streams and dams (Table 6-2).

Table 6-2 Scenario 1: Pre-operational groundwater flow balance.

No	Component	Inflow (Mℓ/d)	Outflow (Mℓ/d)	Balance (Mℓ/d)
1	Dolomite aquifer - Blesbok spruit catchment	10.39	0.00	10.39
2	Karoo and other fractured aquifers - Blesbok spruit catchment	31.68	0.00	31.68
3	Dolomite aquifer - Withok spruit catchment	5.92	0.00	5.92
4	Karoo and other fractured aquifers - Withok spruit catchment	40.11	0.00	40.11
5	Surface water streams and springs (Blesbok Spruit)	5.00	-76.50	-71.50
5.1	Blesbok South	0.00	0.00	0.00
5.2	Blesbok Central	0.00	0.00	0.00
5.3	Blesbok North	0.00	0.00	0.00
6	Ingress areas	0.00	0.00	0.00
6.1	Jan Smuts Dam	0.00	-0.80	-0.80
6.2	Alexander Dam	0.00	-1.50	-1.50
6.3	Cowles Dam	0.00	-2.30	-2.30
6.4	West Pit	0.00	0.00	0.00
6.5	Leeupan	0.00	0.00	0.00
6.6	Van Rhyn Pit	0.00	0.00	0.00
6.7	Largo Colliery	0.00	0.00	0.00
7	Withok Spruit	0.15	-12.00	-11.85
8	Main Reef abstraction GV No 3 shaft	0.00	0.00	0.00
8.1	Sallies Basin Mine	0.00	0.00	0.00
8.2	Brakpan Basin Mine	0.00	0.00	0.00
8.3	Modderfontein Mine	0.00	0.00	0.00
8.4	Central Grootvlei Mine	0.00	0.00	0.00
8.5	Eastern Grootvlei Mine	0.00	0.00	0.00
	Total	**93.25**	**-93.10**	**0.15**
	Imbalance (%)			**0.2%**

6.2.12 Scenario 2: Present day operational phase

From the pre-operational scenario, the present day case with active dewatering was simulated.

The alteration of the subsurface and the surface environment due to mining brought about substantial changes. The main change is the reversal of gradients towards the underground mine void. These changes cause environmental impacts specifically on the surface water and groundwater components. The underground changes are the creation of the mine void and residue facilities that consists of tailings dams and waste rock dumps (Figure 6-14).

6.2.12.1 Scenario 2: System water balance for operational phase

The changes induced by the mine developments are evaluated in this phase. The purpose is to determine the origins of the dewatered mine water that is abstracted from the mine void. Once the origins are established an understanding of the system dynamics are developed. Based on this understanding, management decisions can be identified with possible solutions.

Rainfall: This component is unchanged from the previous phase with a total of 2936.5 Mℓ/d on average received from rainfall on the soil zone. Additional scenarios on climate change could be run with higher or lower annual rainfall rates, this is however kept constant for the purpose of this study (Table 6-4).

Soil: 2845.8 Mℓ/d falls on the soil zone, which is less than during the previous phase as 16.66 km^2 (1.2% of the surface area) of the catchment area is covered with tailings disposal facilities (TDFs) and 8.52 km^2 (0.6 % of the surface area) is covered with mine waste rock dumps (WRD). From the soil zone, most (2586 Mℓ/d) is evaporated and 142.3 Mℓ/d is received by surface water bodies as runoff. A total of 117.5 Mℓ/d infiltrates to the groundwater zone as recharge, which is 33% higher than the previous phase. This is due to increased groundwater gradients created by the mine dewatering as well as seepage from TDFs and WRDs which have a higher groundwater infiltration than the natural system. The higher groundwater infiltration is due to the shape and permeability of the TDFs and the WRDs, which allows higher infiltration rates.

Site surface water: Due to mining activities, the site surface water covers a larger area than during the pre-operational phase. Ponding occurs due to clogging of surface water drainage structures such as culverts below roads. The ponding is exacerbated by erosion and sedimentation from TDFs which causes leaching of contaminants and increased inflow from surface water sources (Figure 6-15). A total of 37.7 Mℓ/d, which has doubled from the previous phase, is received directly from rainfall on surface water bodies and 142.3 Mℓ/d from soil surface runoff. A large component of 47.9 Mℓ/d evaporates and 15.3 Mℓ/d, is received from the groundwater component of base flow. The reduction in the groundwater component of base flow is due to the fact that this component now reports to the mine void.

Sewage Treatment Plant (STP) discharges: The sewage plant discharges accounts for 140 Mℓ/d, which represents 85% of the surface runoff in the pre-operational phase.

Tailings Disposal Facilities (TDFs): The TDFs are important mining features and contaminant sources. The waste from the reef developments are contained in these dumps. The waste material contains contaminants such as sulphur and uranium that originate from the gold bearing reefs. The contaminants are carried with the water to the site surface water, site groundwater and underground mine voids. The nature of the contaminant problem is that the sulphur that leaches from the pyritic minerals (FeS_2) causes acid mine drainage. The TDFs are presently not operational and are considered a very long-term source (i.e. almost constant source) of acid formation and contaminant generation. It receives 35.1 Mℓ/d from rainfall of which 3.5 Mℓ/d is runoff to surface water bodies and 10.5 Mℓ/d seeps to the groundwater zone, while 21.1 Mℓ/d evaporates back to the atmosphere.

Waste Rock Dumps (WRDs): The WRDs cover a much smaller area than the TDFs and also contain less mineralised waste rock material. Although it has an impact, it is lower than the TDFs. The WRDs are also considered long-term sources of contaminants. The WRDs receive 18 Mℓ/d from rainfall of which 0.9 Mℓ/d is runoff to surface water bodies and 5.4 Mℓ/d seeps to the groundwater zone, while 11.7 Mℓ/d evaporates back to the atmosphere.

Underground mine void: The Main Reef was the most important area that was historically mined. It has a calculated open void volume of 170 000 m^3 to 850 000 m^3[45] above the 900 mamsl level (Table 6-5). This volume would still be flooded with rising mine water. The mined out reefs constitute a very long-term source of contaminants with the same minerals and contaminants discussed in the TDF section above. The open mine void was historically completely open and dewatered. Currently, it is flooded to the level of the main pump station at Grootvlei No 3 shaft at 900 mamsl (Figure 5-4, Figure 6-11). The mine void which produces 85 Mℓ/d is by far the greatest contributor of flow and contaminant mass during the operational phase. Its contribution based on flow is 3 times the combined surface water and groundwater contribution from the mine residue facilities (TDFs and WRDs) that amount to 25.7 Mℓ/d.

[45] This range is due to potential differences in porosity, which is an unknown and unknowable parameter (Table 6-5).

Groundwater: Recharge from rainfall of 117.5 Mℓ/d is received via soil infiltration beyond the root zone. The recharge is balanced by the groundwater component of base flow of 15.3 Mℓ/d and 67.2 Mℓ/d is lost to evapo-transpiration in the surface water and groundwater interaction zone. The enhanced groundwater recharge is due to the mine void and dewatering of it.

Table 6-3 Scenario 2: FERB groundwater flow balance – model calibrated

No	Component	Inflow (Mℓ/d)	Outflow (Mℓ/d)	Balance (Mℓ/d)
1	Dolomite aquifer - Blesbok spruit catchment	10.39	0.00	10.39
2	Karoo and other fractured aquifers - Blesbok spruit catchment	31.68	0.00	31.68
3	Dolomite aquifer - Withok spruit catchment	5.92	0.00	5.92
4	Karoo and other fractured aquifers - Withok spruit catchment	40.11	0.00	40.11
5	Surface water streams and springs (Blesbok Spruit)	14.00	-45.50	-31.50
5.1	Blesbok South	6.00	0.00	0.00
5.2	Blesbok Central	4.00	0.00	0.00
5.3	Blesbok North	4.00	0.00	0.00
6	Ingress areas	35.05	0.00	35.05
6.1	Jan Smuts Dam	0.35	0.00	0.00
6.2	Alexander Dam	3.70	0.00	0.00
6.3	Cowles Dam	15.00	0.00	0.00
6.4	West Pit	10.00	0.00	0.00
6.5	Leeupan	4.00	0.00	0.00
6.6	Van Rhyn Pit	1.40	0.00	0.00
6.7	Largo Colliery	0.60	0.00	0.00
7	Withok Spruit	3.00	-9.65	-6.65
8	Main Reef abstraction GV No 3 shaft	0.00	-85.00	-85.00
8.1	Sallies Basin Mine	0.00	-10.00	0.00
8.2	Brakpan Basin Mine	0.00	-5.00	0.00
8.3	Modderfontein Mine	0.00	-34.00	0.00
8.4	Central Grootvlei Mine	0.00	-32.00	0.00
8.5	Eastern Grootvlei Mine	0.00	-4.00	0.00
	Total	**140**	**-140**	**-0**
	Imbalance (%)			**0.0%**

Downstream surface water: The downstream surface water acts as a boundary condition (sink) to the system which receives 404.9 Mℓ/d from the site surface water. This is almost 2.5 times the runoff from the pre-operational phase. The reason is mostly from STP discharges and increased runoff with direct rainfall due to the altered land use. The downstream surface water receives contaminated discharges from the mine void dewatering. The contaminated discharges contain acid water with very high levels of total dissolved solids (TDS) that consists mainly of sulphate and iron. The downstream environment is heavily impacted by these discharges. This is due to the fact that the site surface water and downstream surface water was altered significantly from the pre-operational phase.

Downstream groundwater: The downstream groundwater acts as a sink which receives approximately 6 Mℓ/d from groundwater exchange with the local aquifer.

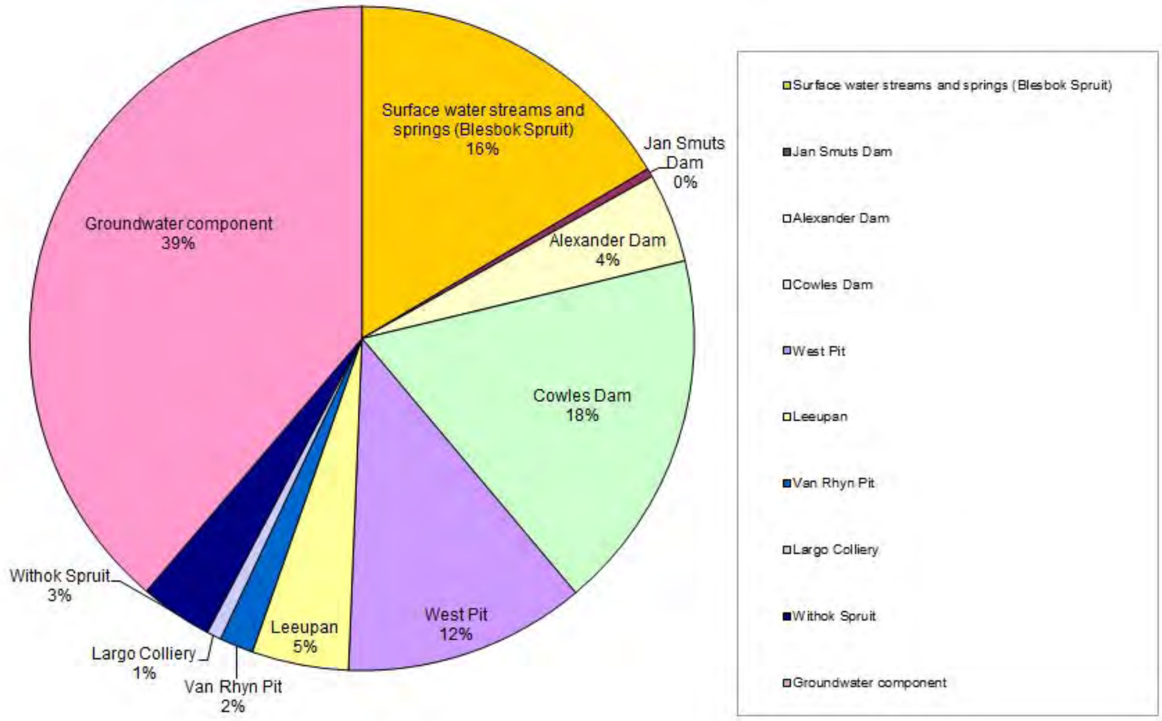


Figure 6-10 Stage 2: FERB: Groundwater inflow components (AGES, 2006).

6.2.12.2 Scenario 2: Groundwater balance

The average dewatering rate at the Grootvlei Mine No 3 Shaft shaft is 85 Mℓ/d (AGES, 2005b). The calibrated model (Section 6.2.9, Figure 2-17) indicated that this volume is possible based on the surface water inflow and the groundwater recharge component within the surface water catchment that was defined as the problem boundaries (Table 6-3, Figure 6-14). The simulated inflow rates are from surface water at 50 Mℓ/d (60%) of which 35 Mℓ/d (40 %) is from ingress[46]

[46] Unnatural high infiltration almost directly into the underground mine void due to shallow mining, is termed ingress.

areas. The ingress zones that are the most important are Cowles Dam 15 Mℓ/d (42 %) and the West Pit canal area 10 Mℓ/d (28 %), which contributes 70 % of the ingress volume of 35 Mℓ/d.

The Blesbok Spruit sub-catchment is the most important and accounts for 70 Mℓ/d (80 %) of the abstracted volume. The present flow in the Blesbok Spruit is due to artificial discharges from sewage works that is in the order of 140 Mℓ/d. The Sallies and Brakpan Basins could account for 15 Mℓ/d (20%) of the dewatered volume, which originates from both the Blesbok Spruit and the Withok Spruit catchments. The Withokspruit accounts for only 3 Mℓ/d (3.5 %) of inflow into the underground mine workings and is therefore much less important.

6.2.13 Scenario 3: Future potential for post-operational mine rewatering and decanting with management and mitigation options

The simulation of the post-operational phase when the FERB will rewater was simulated by cessation of abstraction from the Grootvlei No 3 Shaft to evaluate the potential effects. When pumping ceases, the mine void will flood due to the surface water and groundwater gradients that would exist between the level in the mine void and the sources. The head gradient in the mine void would be relatively flat and it would rise and decant to surface at the lowest point, which is the sub Nigel No 3 Shaft (Figure 6-17, Figure 6-18, Figure 6-19, Figure 6-20). There would also be subsurface decanting from the mine void to the downstream groundwater. This would lead to surface and subsurface decanting of contaminated water to the receiving environment.

For this scenario, it is conservatively assumed (Section 3.7) that no mine rehabilitation is done, but that decanting is not allowed by the regulatory authorities and pumping at a critical base level[47] (CBL) of 1500 mamsl (Figure 6-20) continues to prevent decanting at one of the shafts or below Nigel Central Business District (CBD).

[47] The CBL is defined here as a level that is significantly below the lowest decant point at both the FERB and the CRB, but which would also keep the basins flooded to the extent that oxygen is largely kept out of the system.

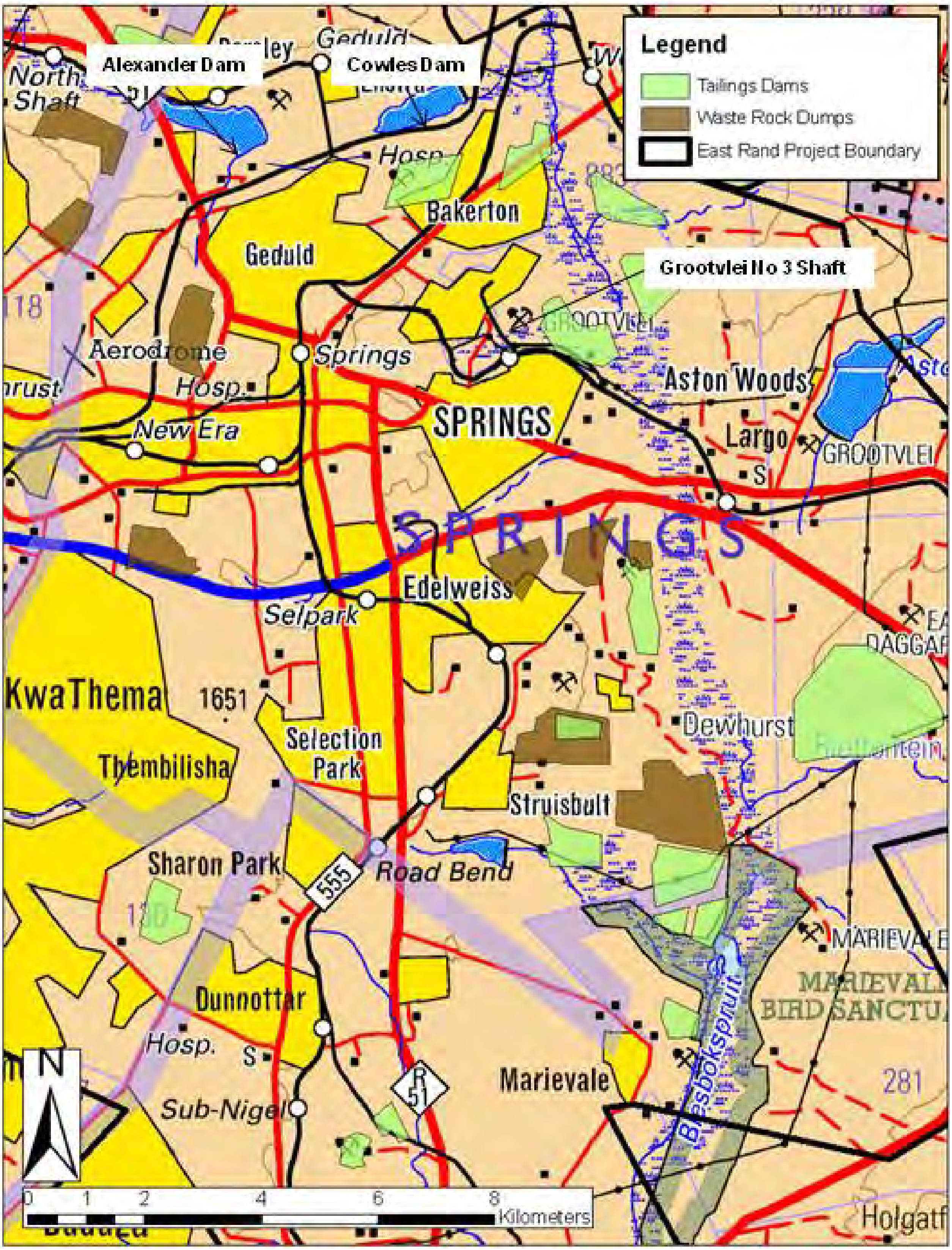


Figure 6-11 Topographic map showing the FERB dams and Grootvlei No 3 Shaft.

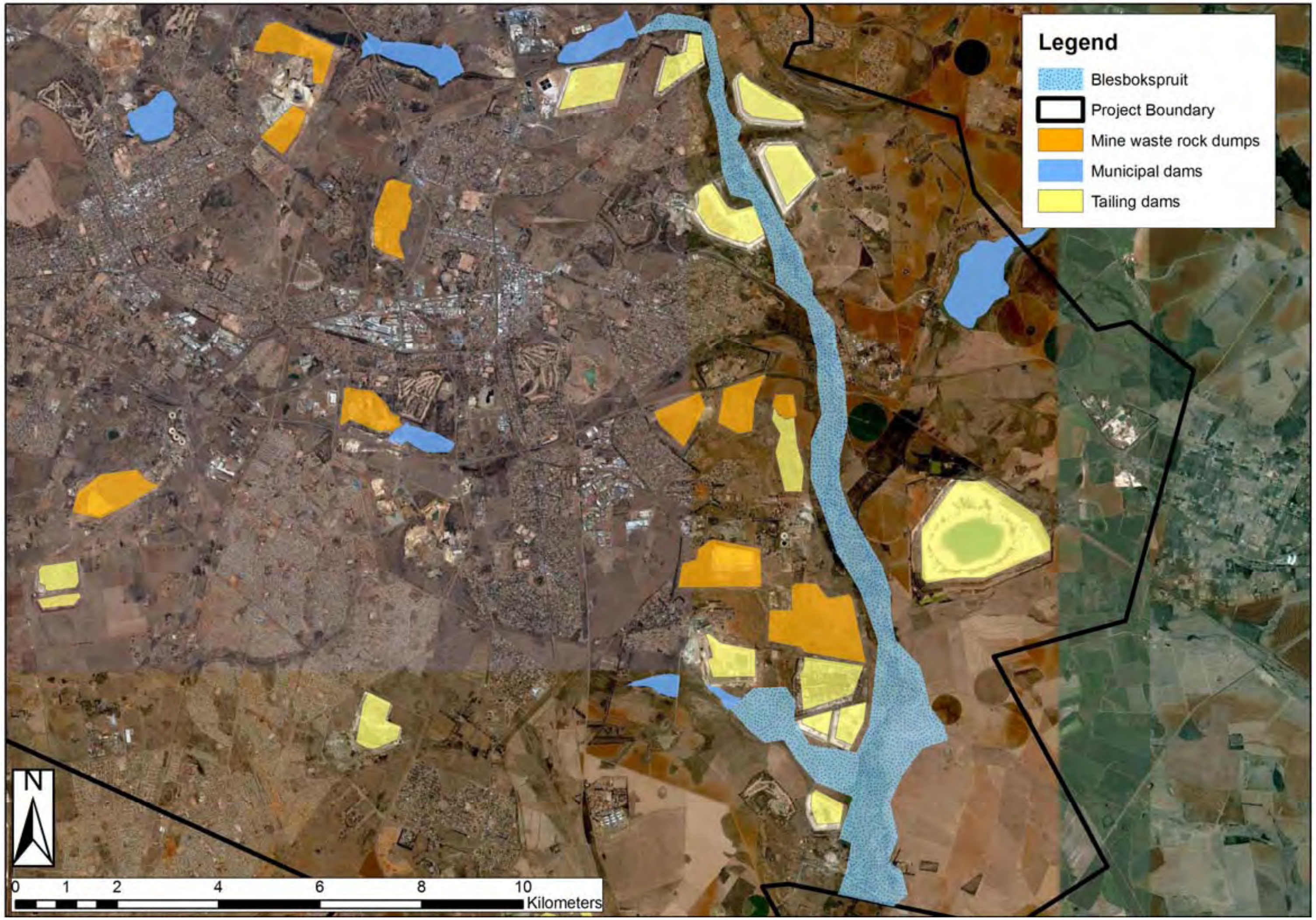


Figure 6-12 Google Earth (Pro) image showing the FERB dams and the Blesbokspruit.

Table 6-4 Scenarios 1 to 3: System water balance (Figure 6-13, Figure 6-14, Figure 6-16).

Components and pathways / processes				Scenario 1: Pre-operational	Scenario 2: Operational	Scenario 3: Post-operational
No	Flow from component	Flow via pathway or process	Flow to component	Flow (ML/d)	Flow (ML/d)	Flow (ML/d)
1	Atmosphere	Precipitation		2 936.5	2 936.5	2 936.5
2	Atmosphere	Precipitation	Soil	2 919.1	2 845.8	2 855.5
3	Atmosphere	Precipitation	TDF	0.0	35.1	35.1
4	Atmosphere	Precipitation	WRD	0.0	18.0	18.0
5	Atmosphere	Precipitation	Site surface water	17.4	37.7	27.9
6	Soil	Runoff	Site surface water	146.0	142.3	142.8
7	TDF	Runoff	Site surface water	0.0	3.5	3.5
8	WRD	Runoff	Site surface water	0.0	0.9	0.9
9	Soil	Infiltration	Groundwater	88.1	117.5	102.8
10	TDF	Infiltration	Groundwater	0.0	10.5	10.5
11	WRD	Infiltration	Groundwater	0.0	5.4	5.4
12	Soil	Evaporation	Atmosphere	2 685.1	2 586.0	2 610.0
13	TDF	Evaporation	Atmosphere	0.0	21.1	21.1
14	WRD	Evaporation	Atmosphere	0.0	11.7	11.7
15	Sewage discharges	Flow	Site surface water	0.0	140.0	140.0
16	Site surface water	Evaporation	Atmosphere	22.2	47.9	35.5
17	Site surface water	Flow	Downstream surface water	163.4	404.9	360.7
18	Groundwater	Flow	Losses (ET)	70.5	67.2	71.8
19	Site surface water	Ingress	Mine voids	0.0	50.0	35.0
20	Groundwater	Seepage	Mine voids	0.0	35.0	15.0
21	Mine voids	Dewatering	Site surface water	0.0	85.0	50.0
22	Upstream groundwater	Flow	Groundwater	1.0	1.0	1.0
23	Groundwater	Flow	Downstream groundwater	11.0	6.0	8.0
24	Groundwater	Base flow	Site surface water	17.6	15.3	16.0

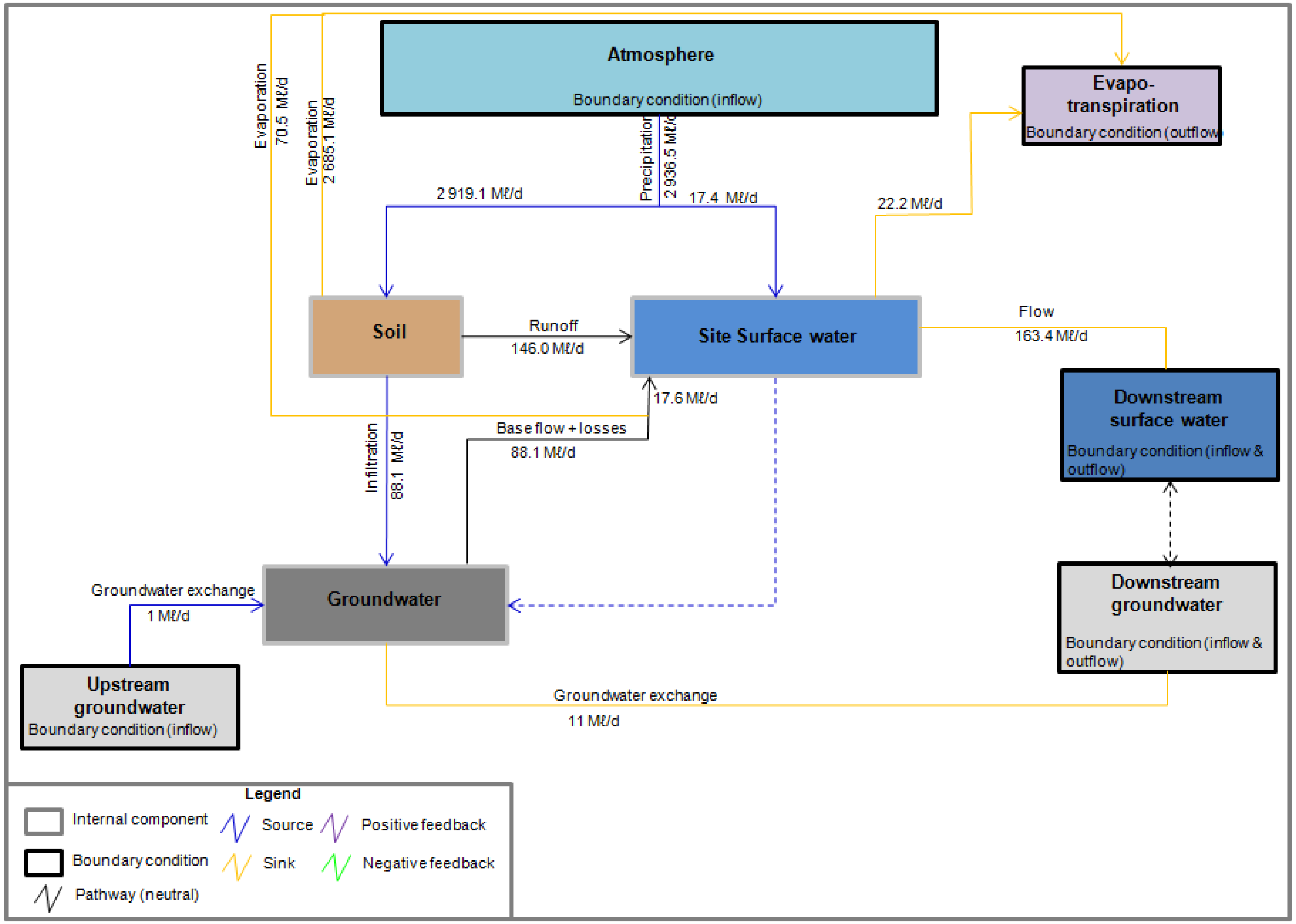


Figure 6-13 Scenario 1: Conceptual process flow model with average daily flow rates during the pre-operational phase.

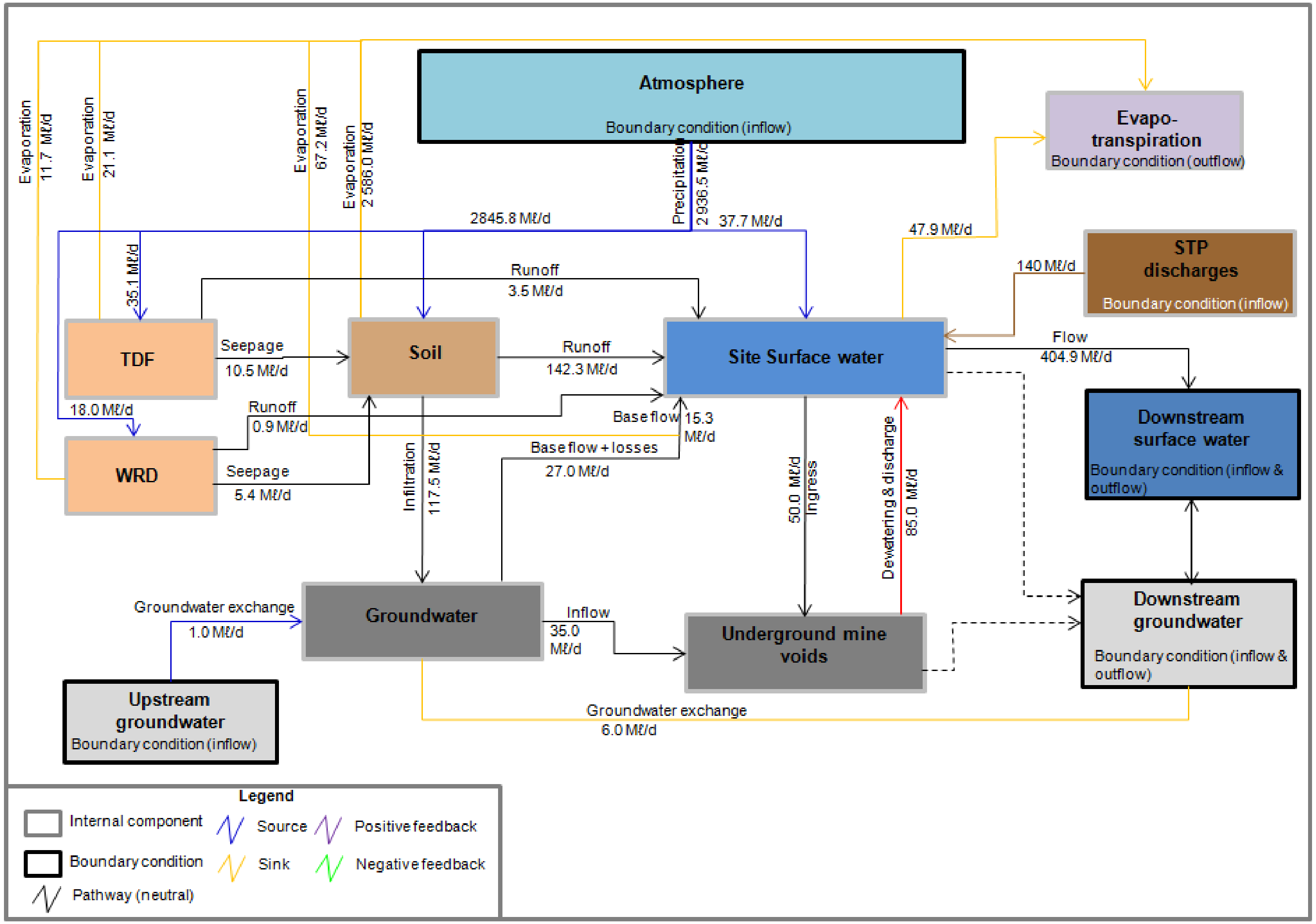


Figure 6-14 Scenario 2a: Conceptual process flow model with average daily flow rates during the operational phase.

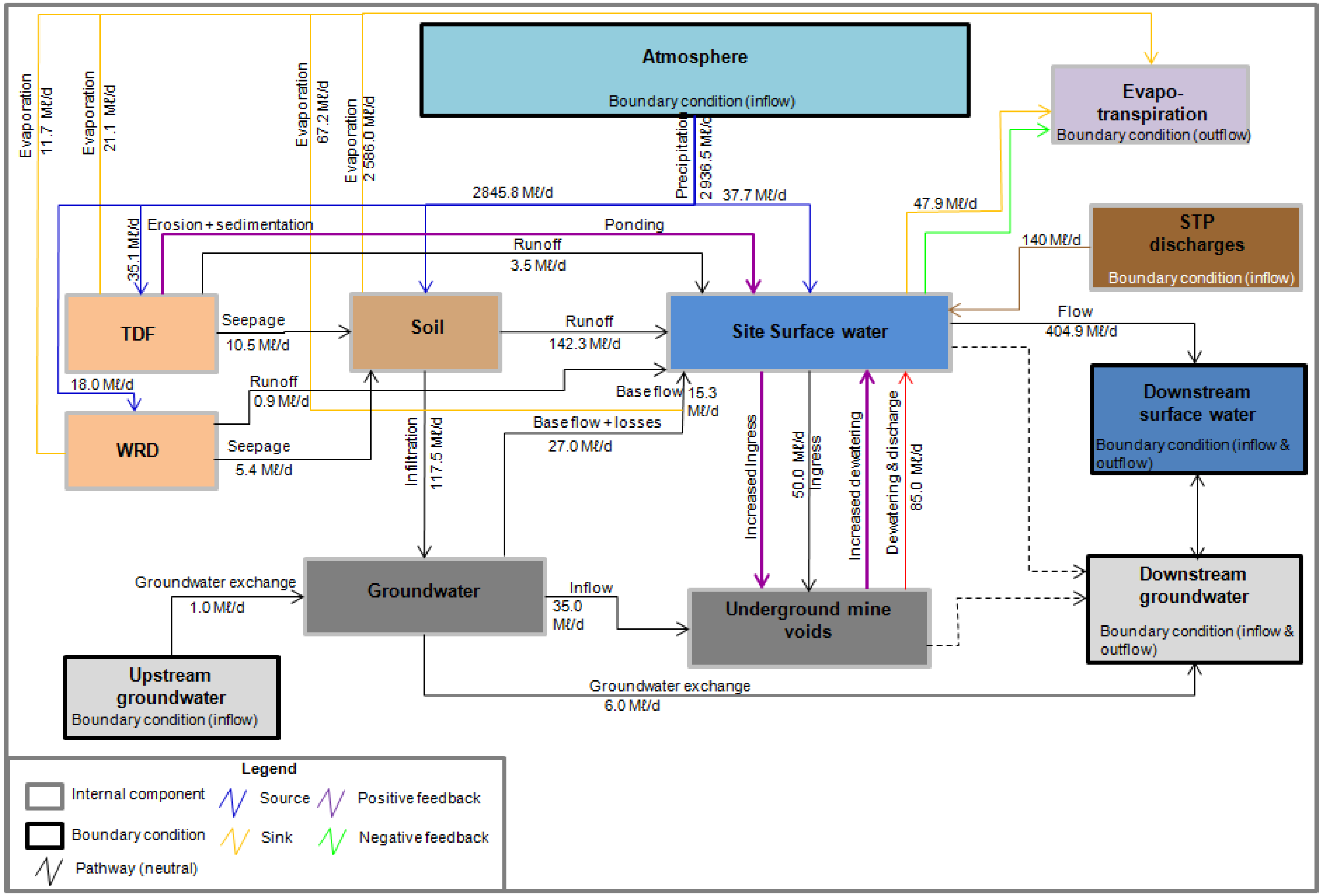


Figure 6-15 Scenario 2b: Conceptual process flow model with average daily flow rates during the operational phase – showing the effect of erosion and sedimentation.

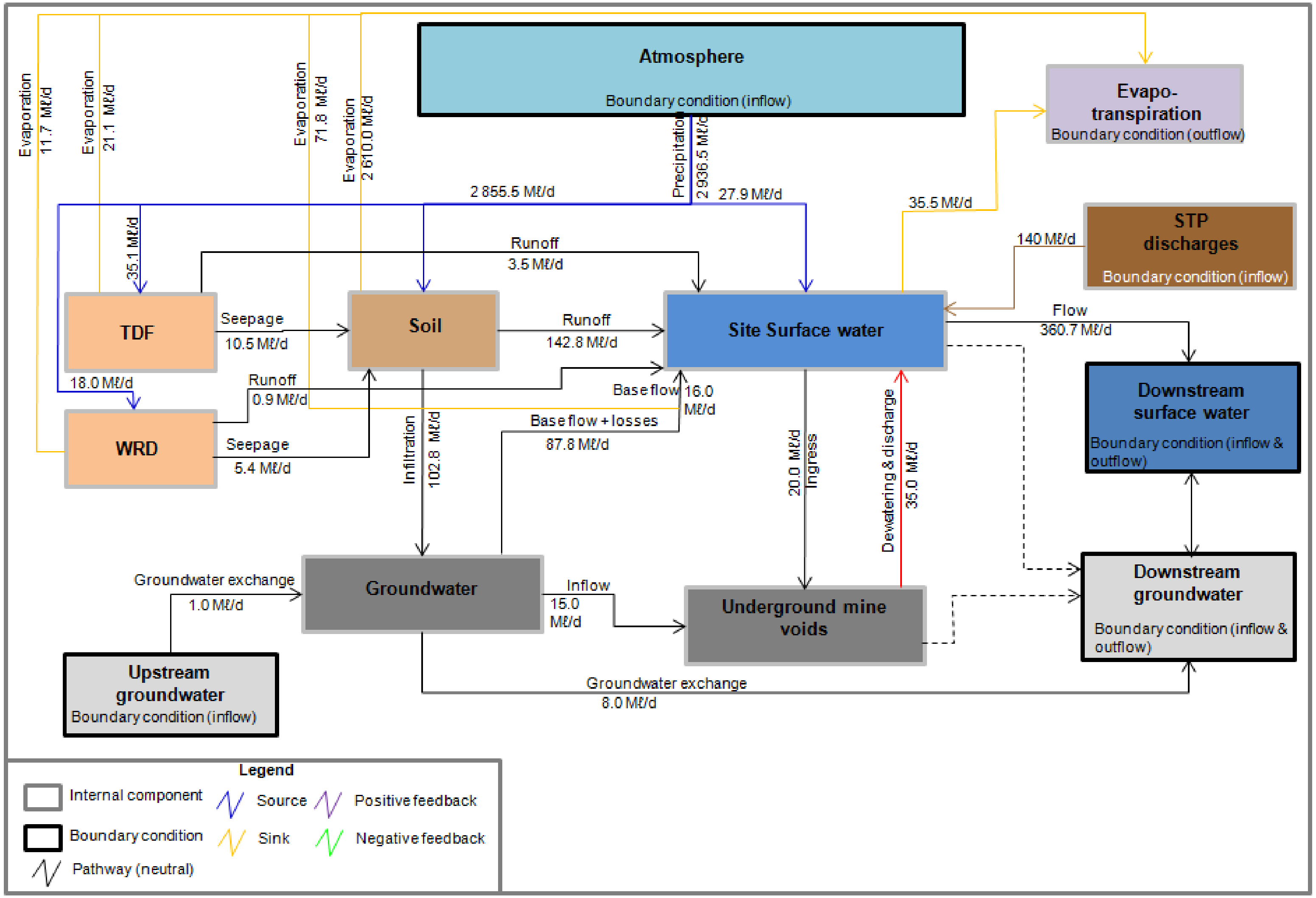


Figure 6-16 Scenario 3: Conceptual process flow model with average daily flow rates during the post-operational phase.

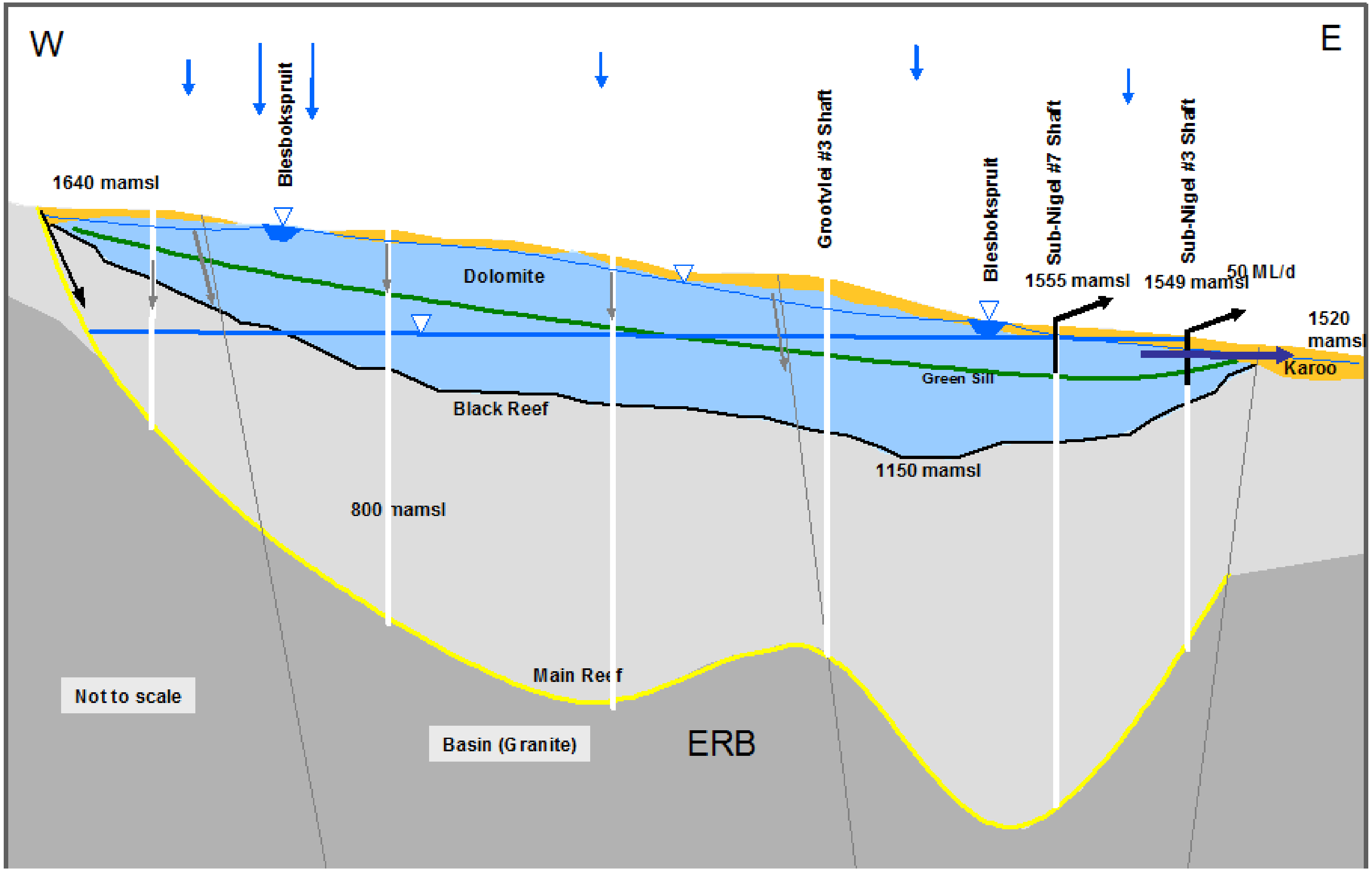


Figure 6-17 Scenario 3: Conceptual model for post-operational decanting and sub-surface seepage at Sub-Nigel #3 Shaft.

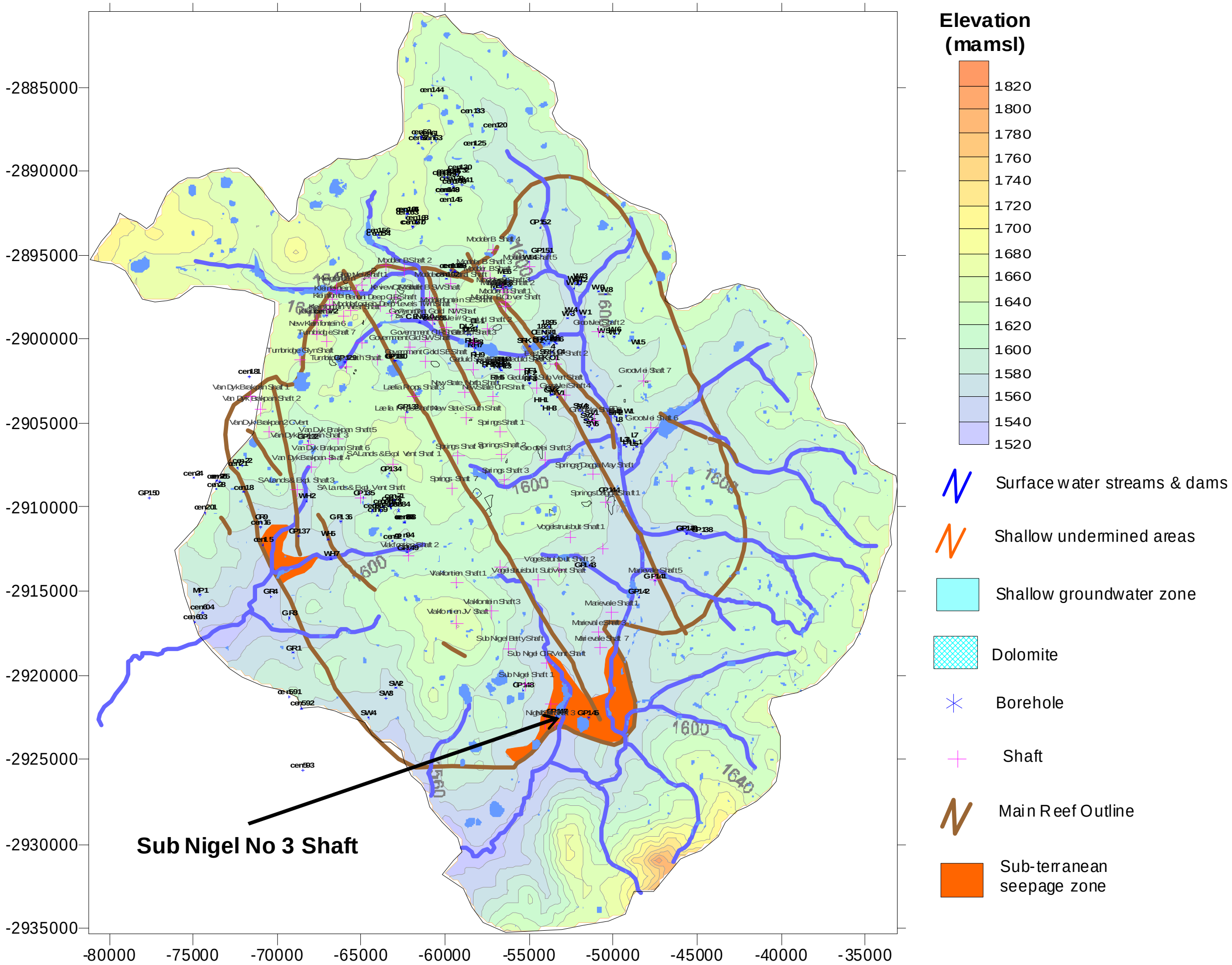


Figure 6-18 Scenario 3: Post-operational decant and sub-surface seepage zones.

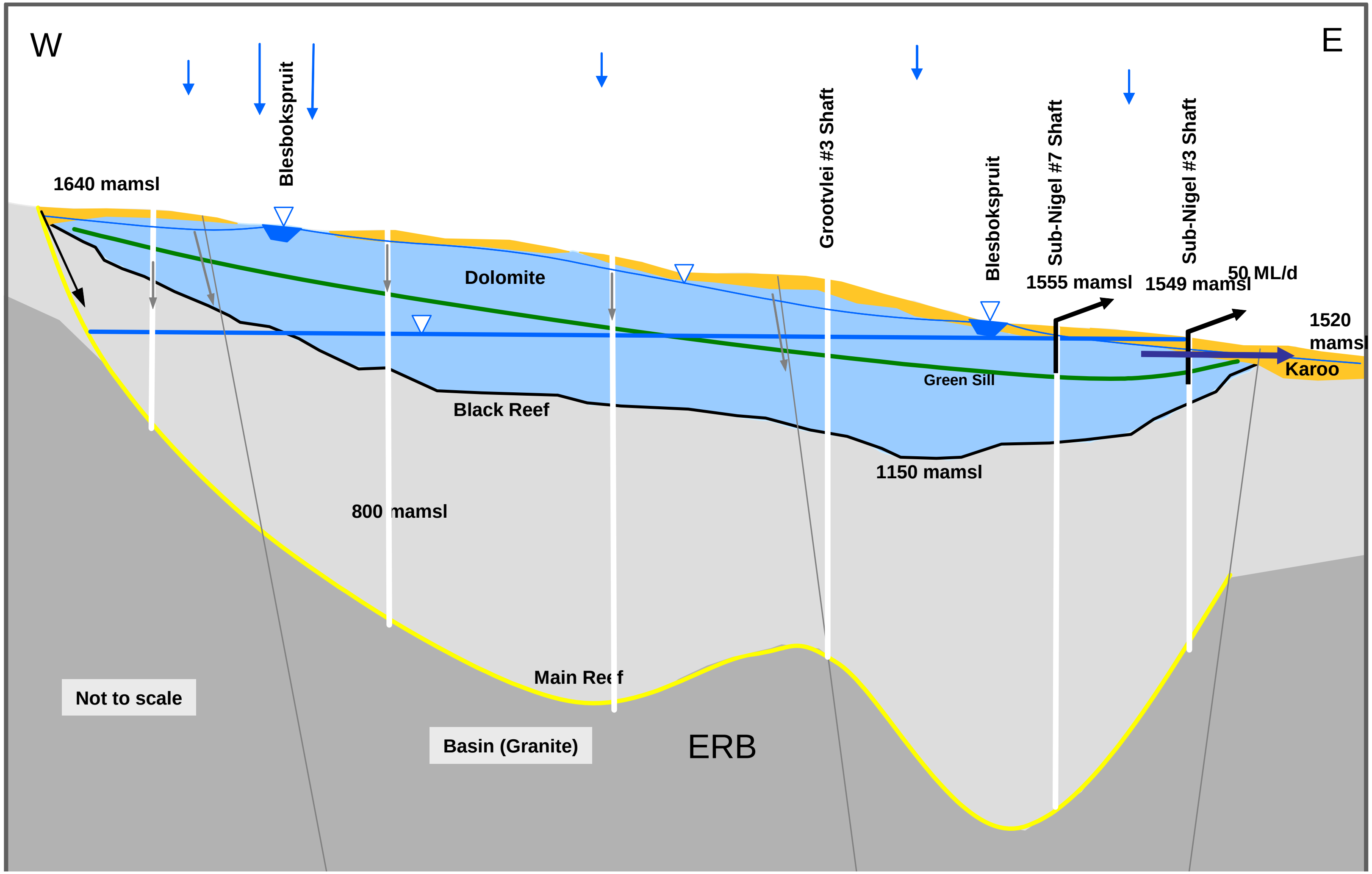


Figure 6-19 Scenario 3: Post-operational decanting and sub-surface seepage at Sub-Nigel #3 Shaft (AGES, 2006).

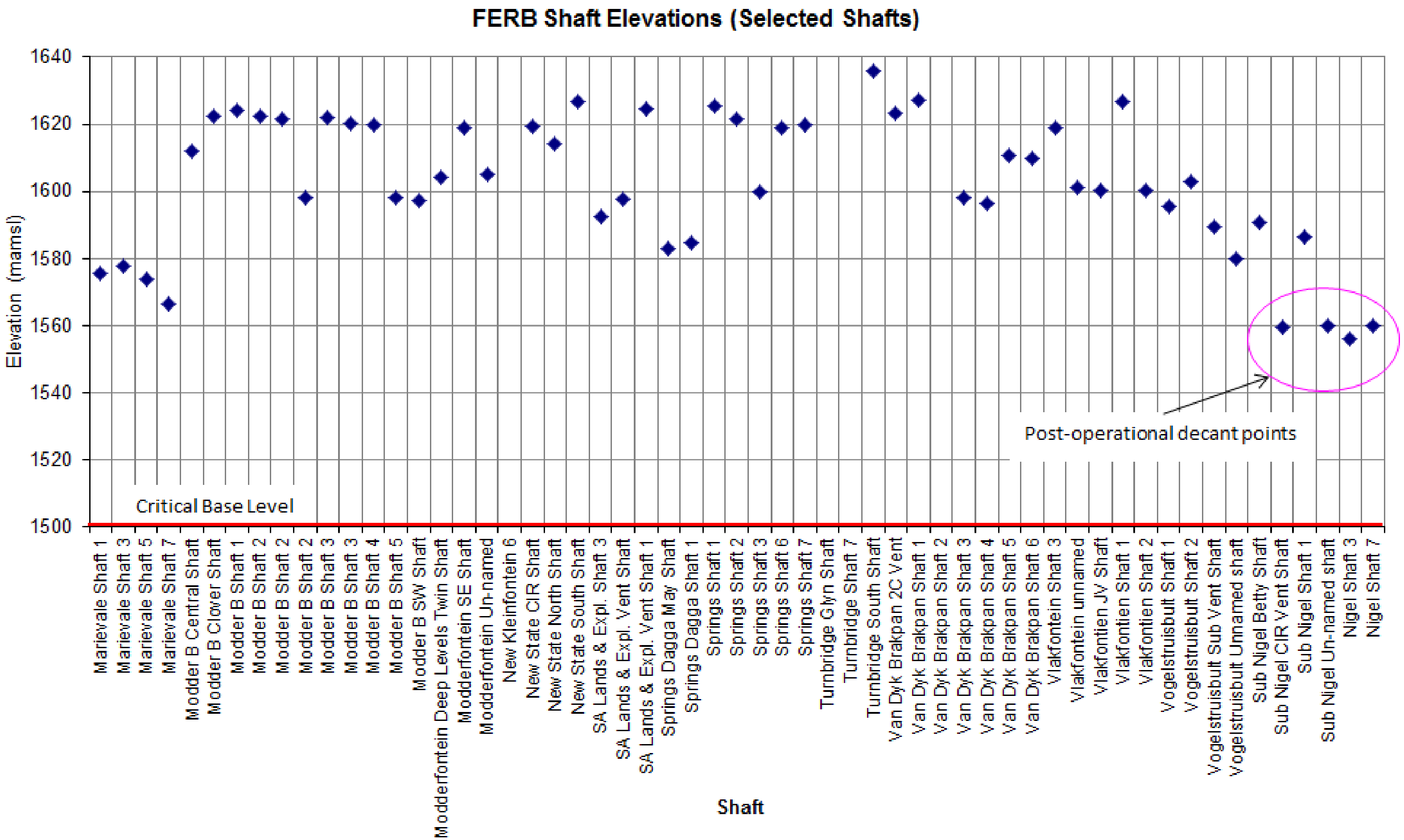


Figure 6-20 Scenario 3: Post-Operational decanting – selected shaft elevations (AGES, 2006).

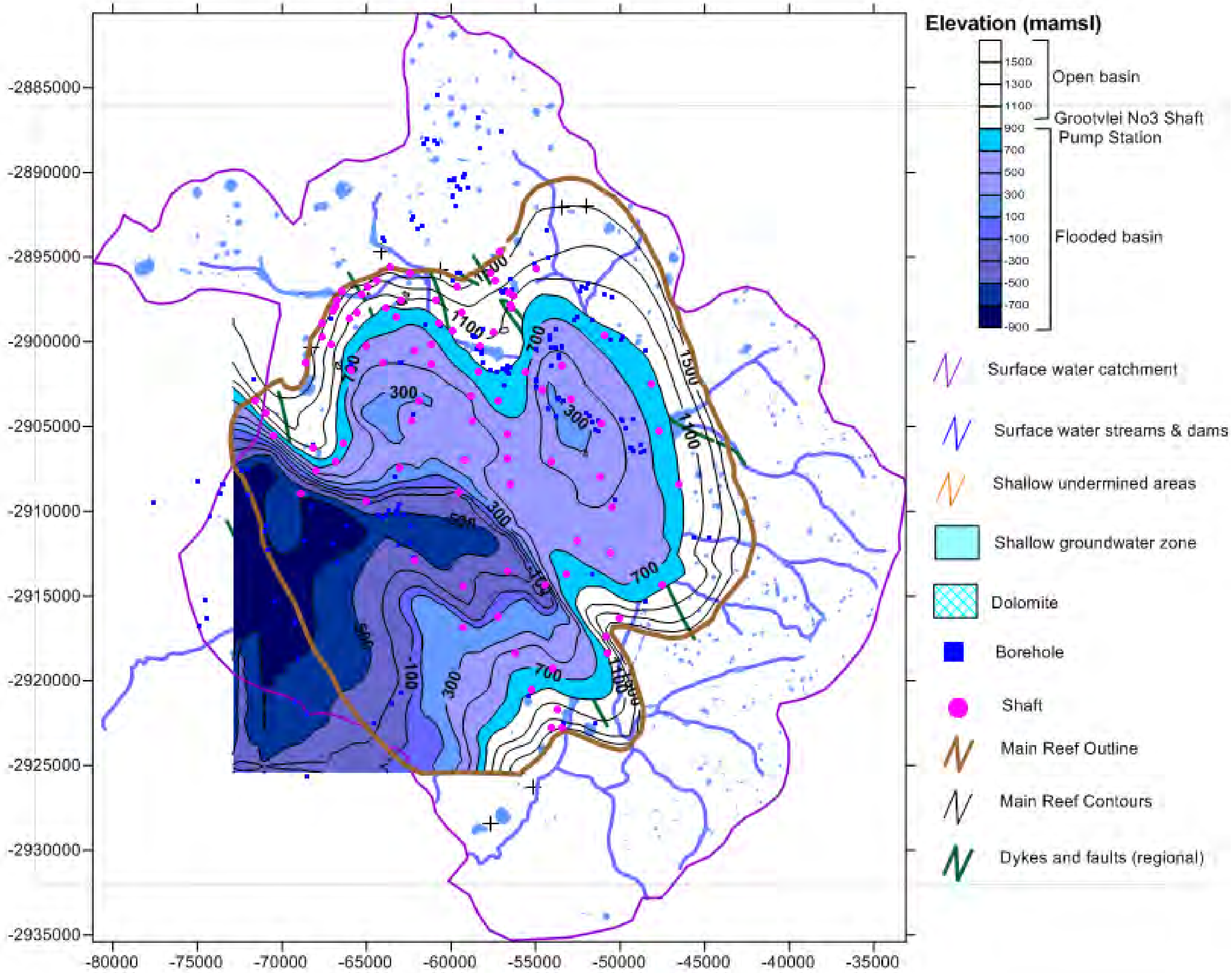


Figure 6-21 Scenario 3: Study area showing the extent of the mined out area in Main Reef (AGES, 2006).

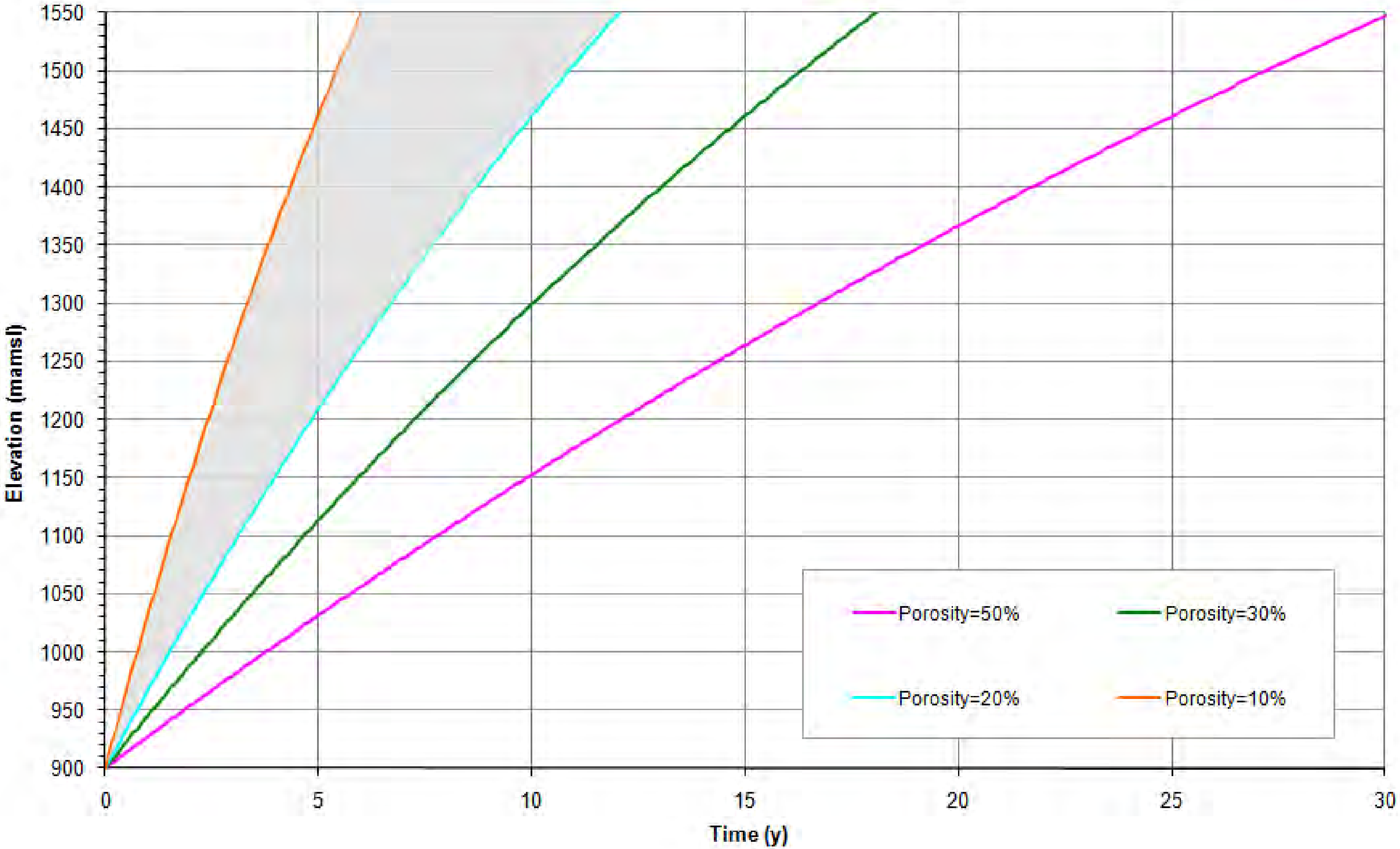


Figure 6-22 Stage 3a: Time dependant rewatering rates at varying porosity values for Main Reef.

6.2.13.1 Scenario 3: System water balance for the post-operational phase

Rainfall: Like in the previous phases, rainfall is unchanged from the previous phase with a total of 2936.5 Mℓ/d on average received from rainfall on the soil zone (Table 6-4, Figure 6-16).

Soil: 2855.5 Mℓ/d falls on soil zone, which is more than during the operational phase as the area covered by surface water would shrink. This is due to lower dewatering and hence smaller surface water ponding areas. From the soil zone, most (2610 Mℓ/d) is evaporated and 142.8 Mℓ/d is received by surface water bodies as runoff. A total of 102.8 Mℓ/d infiltrates to the groundwater zone as recharge, which is lower than the operational phase. This is due to decreased groundwater gradients created by the mine dewatering. The impact from the TDFs and the WRDs would be the same as the current operational phase as these facilities are not operational at present. Although dewatering takes place, the (Aurora) mines are on care and maintenance.

Site surface water: Due to the cessation of the mining activities and hence dewatering that is reduced, the site surface water covers a smaller area than during the operational phase. Ponding still occurs due to clogging of surface water drainage structures such as culverts below roads, but is now less significant than during the operational phase when larger dewatering volumes were discharged. A total of 27.9 Mℓ/d, which is 25% less than the previous phase, is received directly from rainfall and 142.8 Mℓ/d from soil surface runoff. A large component of 35.5 Mℓ/d evaporates and 16 Mℓ/d is received from the groundwater component of base flow. The slight increase in the groundwater component of base flow is due the fact that groundwater inflow and hence dewatering is reduced.

Sewage Treatment Plant (STP) discharges: The sewage plant continues to discharge at 140 Mℓ/d, which is still 85% of the surface runoff in the post-operational phase.

Tailings Disposal Facilities (TDFs): The TDFs are not rehabilitated. The ingress of oxygen and enhanced leaching of sulphide minerals exacerbates the acid mine drainage problem. The water quality that seeps to the underlying soil and the runoff to surface water is much worse than during the operational phase. The status is the same as in the operational phase.

Waste Rock Dumps (WRDs): The WRDs are also not rehabilitated, but are not expected to have a higher or lower impact than during the operational phase as the status is the same.

Underground mine void: The underground void is flooded to the BCL of 1500 mamsl, which is 56 m below the elevation of the lowest shaft Sub-Nigel No 3 Shaft (Figure 6-19, Figure 6-20). This shaft is located at an elevation of 1556 mamsl. The head is maintained at 1500 mamsl by pumping. Due to the flooding of the mine void, the head gradient is decreased with groundwater inflow and surface water inflow that also decreases to 50 Mℓ/d (15 Mℓ/d from groundwater and 35 Mℓ/d from surface water). The effect of flooding of the basin reduces the ingress of oxygen significantly and the water quality of the dewatered component increases significantly. The acidity is lower and the sulphate and iron concentrations are also reduced. The dewatering of the mine void of 50 Mℓ/d is still the most significant factor in terms of flow (not necessarily mass) which is almost double the cumulative flow contribution of the TDFs and the WRDs of 20.3 Mℓ/d.

Groundwater: Recharge from rainfall of 102.8 Mℓ/d is received via soil infiltration. This is less than during the operational phase, due to the smaller cone of depression and shallower head gradients of the partially flooded basin. The recharge is balanced by the groundwater component of base flow of 16 Mℓ/d and 71.8 Mℓ/d that is lost to evapo-transpiration in the interaction zone.

Downstream surface water: The downstream surface water acts as a boundary condition (sink) to the system which receives 360.7 Mℓ/d from the site surface water. This is a small reduction of 11% from the operational phase. Unless corrective measures are taken, the downstream surface water will continue to receive contaminated discharges from the mine void dewatering indefinitely. The contaminated discharges contain acid water with still high levels of total dissolved solids (TDS) that consists of mainly sulphate and iron that would have a significant impact on the downstream environment.

Downstream groundwater: If no corrective measures are taken, the flooded mine void water will seep into the downstream groundwater indefinitely. The seepage of 8 Mℓ/d would mainly take place via preferential pathways such as faults and dykes and through the upper, weathered zone. The contaminated seepage would decant as seepage zones or springs below the Nigel CBD and surrounding areas (Figure 6-18).

6.2.13.2 Scenario 3: Groundwater balance for post-operational phase

The simulations indicated that due to regional topographic elevation differences, there would be a relatively flat piezometric gradient of 5E-04 (or 1:2000) and that the head elevation *will not be* *perfectly horizontal* as was previously believed. This is due to the fact that the open mine void does not constitute unconstrained flow as there are areas where reef collapse cause water to follow alternative routes emulating macro porous flow. It would mean that decanting that was expected at only one point, Sub-Nigel No 3 Shaft, which is at the lowest elevation of 1556 mamsl, could also occur at other shafts. The relatively flat, but still curved piezometric surface, together with the flat topography of the FERB causes the decanting to take place from a number of shafts and to the Blesbok Spruit. These are; Sub-Nigel Shafts No 3, No 1, No 7 and Sub-Nigel CIR Vent Shaft (Figure 6-18, Figure 6-20). The elevation difference between these shafts is small with Sub-Nigel No 3 Shaft at 1556 mamsl and Sub-Nigel No7 Shaft 1560 mamsl (Figure 6-19). These two shafts are located 570 m apart. Sub-surface seepage through the weathered dolomite zones takes place towards the Blesbok Spruit through a seepage zone (Figure 6-19). The simulation indicates that some of the decant water may take place below the Nigel CBD (Figure 6-18).

The total decant volume would be approximately 50 Mℓ/d of which 10 Mℓ/d could be subsurface seepage. The 40% decrease in decant volume compared to the 85 Mℓ/d is due to a decrease on the head gradient, which causes less surface water ingress and groundwater inflow.

Depending on the permeability of the shaft sidewalls, the subsurface seepage could take place over an area of 5 km^2 to 10 km^2. The seepage would take place through the upper 30m to 50 m of weathered Karoo Sandstone (Vryheid Formation) and Alluvium along the Blesbok Spruit. The decant water could form springs along the north-south striking regional fault zone to the east of Sub-Nigel No 3 Shaft.

6.2.13.3 Scenario 3: Modelling of time scales to rewater and decant

The time available since cessation of pumping to decant depends entirely on the open volume of the mined out area between 900 mamsl and 1550 mamsl (Table 6-5). The time scale is important as it would provide the basis for decision-making to determine how the problem can be managed and mitigated and how much time would be available, should the pump station be flooded.

The unsaturated or void area of Main Reef above 900 mamsl covers an area of 2.10E+06 m^2, with a calculated volume of 170 000 m^3 to 850 000 m^3 (Table 6-5, Table 6-6, Figure 6-21). The rewatering was determined by calculating the time that it would take to fill each meter with a linearly decreasing inflow with an increase in elevation, based on Darcy's Law (Equation 2-4, Bear, 1979). The range in volume is influenced by the porosity, which is a very sensitive parameter in the rewatering model. The problem here is that this critically important parameter is unknown. Porosity can be determined by field tracer tests, but in this case, there are large open areas formed over a very large regional scale, which makes it impossible to determine porosity, especially if the spatial variation is taken into account (Section 4.3). The porosity is influenced by the fact that not the whole of Main Reef was mined out and subsidence or collapse of mine support material decreases the void volume. It is not possible to determine the porosity as this parameter depends on the mine void volume still open as most of the 0.8 m stopes is expected to be closed under the overburden weight.

What do we do now that we are confronted with a critically important parameter that is not determinable? The only solution is to make assumptions using the most accurate information available and to qualify these assumptions with calculations or models using sensitivity analysis. The porosity of the mine void space was calculated based on the known surface area of the Main Reef above the 900 mamsl level (Figure 6-21). This uncertainty was quantified by using a sensitivity analysis on the porosity of the Main Reef (Table 6-6).

Table 6-5 Main Reef and decanting data

Description	Quantity
Main Reef Stope Thickness (m)	0.8
Main Reef Area (m^2)	2.13E+06
Main Reef Volume (m^3) at 80 % porosity	1.71E+06
Start level (mamsl)	900
Decant level (mamsl)	1545
Initial inflow rate (m^3/d)	8.50E+04

The rewatering calculations indicated that it could take between 5 to 25 years for the FERB to rewater and decant, based on porosity values between 10% to 50 % and a reef thickness of 0.8 m (Figure 6-22, Table 6-6). This assumption on porosity of 10% to 20 % yields an average open reef of 8 cm to 16 cm across the unflooded part of the basin (Figure 6-21). If the porosity is e.g. 30%, which is unlikely due to compaction in Main Reef, the FERB could take up to 15-18 years to flood and decant.

Table 6-6 FERB rewatering rates and scenarios

Main Reef Porosity (%)	80	50	30	20	10

Effective opening (cm)	64	40	24	16	8
Main Reef unsat void volume (m^3)	1.36E+06	8.53E+05	5.12E+05	3.41E+05	1.71E+05
Rewatering rate (m/d)	0.05	0.08	0.13	0.19	0.38
Rewatering rate (m/month)	1.4	2.3	3.8	5.6	11.3
Rewatering rate (m/y)	17	27	45	68	136
Time to rewater (y)	38	24	14	9	5

The water level rate of rise is calculated at between 5 m to 11 m per month (70 m/year to 135 m/year). The uncertainty with regard to the porosity can be reduced by recalibration of the model with observed data. This is iterating back to Step 4 (Collect and Analyse Data) in the modelling process (Section 5.6, Figure 5-8). Observed data can be obtained by switching off the pumps for say a week or a few days to measure the rewatering rates.

The lower bound for porosity is conservatively assumed to base the management decision-making on (Section 3.7). Based on this assumption, there are 5 to 12 years to implement management and mitigation measures. This way, if the assumption is wrong and the porosity is higher, there is no adverse problem created than to take a view that the porosity could be higher and then have no mitigation system in place. This approach lacks scientific data but provides sufficient information to base management decisions on. The information that is provided shows that action has to be taken immediately to put mitigation and management measures in place, as 5 years is not a long time to plan and implement these measures.

6.2.13.4 Scenario 4: Decision-making on options for management and mitigation

Based on the technical investigations of the current and potential future scenarios on the FERB, several management and mitigation measures or solutions were identified (AGES, 2006). Now that the system boundaries are determined with the components and importance of components and interactions or pathways, options can be evaluated for management and mitigation. It is analogue to game theory (such as a chess game; Section 2.9.6) where several options (or moves) can be evaluated and tested as sub-scenarios, which is similar to sub-systems (Figure 5-2). The identification of constraints in the system is very important, as it could dictate the performance of the system (Goldratt, 1990).

Following the evaluation of management options (Figure 2-1) from the existing components, innovative decisions (Section 2.3.3) have to be made for selected scenarios (Figure 2-2). This is done by identifying additional or potential options that do not yet exist.

The management aspects are characterised from a systems viewpoint that require a holistic and integrated surface water, ground water, mining and broader environmental management perspective. The options for management and mitigation must be based on the principle of sustainability (Section 2.5). A management strategy (Section 2.3.1) is required that takes all the important principles into account and that would obtain mitigation measures that adhere to the sustainability principle.

6.2.13.4.1 Strategic decision- making on management options

The possible solutions to the mine rewatering and decanting problem should be based on defined principles before the technical details can be considered. The objective of the management and mitigation measures or solutions should be to minimise environmental impacts and costs. For this purpose, *some optimization* will be required within the constraints of sustainability parameters (Section 2.9.7). The bounded rationality model for decision-making is followed as it is accepted that all possible options[48] for management cannot be evaluated within the economic and temporal limitations of the project, while *sufficient* (Section 3.5) measures can be taken to reach the stated objectives.

Regulatory: The regulatory framework requires that (i) pollution of water and the environment should be mitigated, (ii) no burdens should be put on future generations because of our actions today and (iii) the polluter pays principle is applicable. The guidelines set within these regulations require that the Best Practicable Option (BPEO) and the Precautionary Principle should be followed (Section 2.10).

Principles: Important requirements are that management measures must be sustainable and environmentally responsible. Options that are not sustainable defer the problem to future stakeholders. The definition of sustainability (Section 2.5) should be included in all the options that are considered, which considers environmental, technical and social (economics, legal, political) aspects. This is where ongoing pumping becomes a very difficult option because it cannot be regarded as sustainable for future generations. Alternatively, a project that allows for pumping of an interim period of e.g. 5 to 10 years should make provision for natural decant

[48] This will require a lot of research that is currently in progress to finish, which is unlikely to happen as the outcomes usually spawn more research.

options as part of the final closure plan. Interim pumping would however be good if it can be used to keep the water level in the basin on different levels to determine the optimum level before permanent infra-structure such as tunnels that are more expensive are considered.

Integration (holism): If the principle of integration is considered, then the problem should be centralised to have one management point and one management option. This option takes advantage of the economy of scale principle and it is easier to manage one point than e.g. a number of possible decant areas. This option is proposed for both the FERB and the CRB by linking the two basins (Scenario 4.4).

Differentiation: It is the opposite of the integration option where the problem is isolated in smaller areas e.g. more than one decant point in the CRB and separate decant for the FERB. This option is not regarded as a practical solution.

Elimination of uncertainties: When dealing with a complex, regional environmental problem, it is proposed to eliminate uncertainties rather than try to manage and quantify them. There are basically two management options. The one is to seal all the ingress areas and the other to pump from the mine void. If it would be practically possible, it would have been better to seal all ingress areas as it would keep clean water clean and reduce the dewatering rate. The problem is that the ingress problem is a diffuse one and sealing is very expensive. The risk is that due to the diffuse nature of the ingress areas, it is not possible to determine or know where these zones are. The known ingress points like open shafts should be sealed, but the focus should be on the known factors such as that the basins will decant. Decant control must become the focus and not the sealing of all ingress areas as most of the ingress areas are largely unknown and it would not be possible to determine the relative contributions of each of the potential ingress areas. It is technically complex and in some cases impossible to identify and seal all the ingress areas. The decision here is to minimise uncertainties and rather manage the known part of the problem.

6.2.13.4.2 Management and mitigation measures

Based on the strategy developed, the following management and mitigation measures or solutions are identified (Table 6-8):

1. Option 4.0 - Do nothing: This option is unlikely but is used as a reference point or base case against which the other options can be compared.
2. Option 4.1 – Operational phase continue: In this option, a scenario is considered where

the future gold price rises to the extent that it is economically viable to reopen the mines.

3. Option 4.2 - Post-operational pumping: The post-operational management of the rising water is managed with pumping based on the current practice. The water level will be kept to keep the basin as full as possible but below the BCL to keep oxygen out as far as possible, which would reduce the effects of AMD. A new long-term pump station will be developed and equipped. A water treatment plant of 40 Mℓ/d would be constructed at the decant point.
4. Option 4.3 - Post-operational natural decant via FERB tunnel: A 25 km long tunnel from Sub-Nigel No 3 Shaft (Figure 6-24) will have to be developed to the BCL at 1500 mamsl to serve as a permanent and sustainable decant point in the FERB. This will allow for natural decant under gravity with no need for pumping. A water treatment plant of 40 Mℓ/d would be constructed at the decant point
5. Option 4.4 - Post-operational natural decant via FERB & CRB link tunnel with decant tunnel: In this scenario, the FERB will be linked with the CRB via a 3 km smaller tunnel at 200 m to 400 m depth. A 35 km tunnel will be constructed from ERPM shaft towards the south to the BCL at 1500 mamsl to serve as a permanent and sustainable decant point in the FERB and the CRB (Figure 6-25). This will allow for natural decant under gravity with no need for pumping. A large water treatment plant of 55 Mℓ/d would be constructed at the decant point.

These options will be evaluated in the next section and tested against the developed strategy of sustainability and principles that was identified.

6.2.14 Scenario 4: Influence of business (economics) and social (legal) aspects on the environmental liability

The simulations and interactions between components for the previous scenarios do not fully qualify as a systems approach, as it does not take all the variables into account that influence sustainability (Section 2.5). There are important factors of business (economics) and social (legal and political) that can influence the future management options that are required in the decision-making process (Section 2.6, Section 2.8).

6.2.14.1 Scenario 4a: The effects of business (economics)

While mining was the driving force behind the development of Johannesburg and the establishment of a sustainable economy in South Africa, the origin of the current problem is driven by economics. When the gold price and cost of mining allowed ongoing mining the water was pumped and treated by large mining companies that had sufficient capital to manage the operational costs of pumping and water treatment.

Background

Gold mining is active in South Africa for 126 years. It started in 1884 and peaked at a production rate of 1 million kg/year in 1970 (Figure 6-23; Hartnady, 2009). The FERB is the largest gold sub-basin in the Witwatersrand Basin. Up to 1995, a total of 46.21 million kilograms of gold was recovered, of which 9.91 million kilograms originated from the FERB, which represents 21.5% of all the gold mined in South Africa up to that time. The average yield in the FERB was high at 8.19 g/t, which was the fourth highest after the West Wits Line, Free State and Klerksdorp basins (Handley, 2004). The total value of gold mined in South Africa calculated at today's price[49] exceeds US$ 2 178 billion. Gold mining in the FERB based at present value would exceed US$ 467 billion or R 3 234 billion. The FERB therefore played its part in the economic development of South Africa, which is an important aspect for consideration. There are still ate least 5000 tonnes of minable gold in South Africa, which makes it still one of the largest future potential gold producers (Hartnady, 2009).

Since 1995, there was a steady decline in gold production in the FERB as mining reached deeper levels and the gold price fell relative to the mining costs. Although there are still reserves left, this made gold mining largely unprofitable and sections in the FERB started to close down. The Grootvlei No 3 Shaft, which was operated by Pamodzi Gold, was the *last man standing* and had to pump all the water that accumulated in the basin. Pamodzi was liquidated in the recession of 2008/9 and the assets were bought by Aurora Mining who did not have the capital to operate or dewater the mine.

[49] The gold price on 25 October 2010 is 1335 US$/ounce and the rand: US$ exchange rate at 6.923. (www.BusinessDay.co.za).

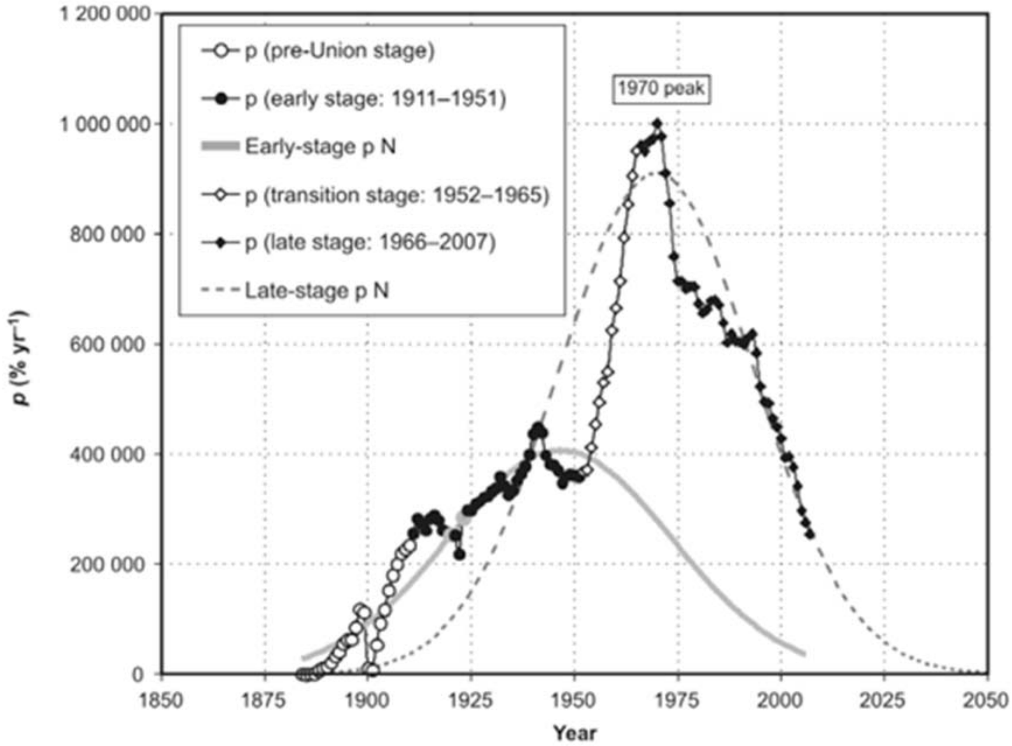


Figure 6-23 History of South African gold mining production rate (p = kg/year) (Hartnady, 2009).

The state received taxes on mine revenue during the economical phase of the mines but during the post-operational phase, this meant that dewatering of derelict or bankrupt mines would largely become the liability of the state. This is because a number of the old mine operations ceased more than 20 to 30 years ago when environmental legislation was not in place. Those that are still idle do not have the capital to manage the environmental liabilities such as ongoing dewatering and treatment. It is important to determine the magnitude of this liability as part of the financial component of sustainability.

Financial assessment

The management of the post-operational options would amount to social and political pressure with associated legal costs for option 4.0 with technical and management costs for Options 4.1 to 4.4. To be able to make a management decision, the decision-maker needs to determine the comparative costs of the respective management options for the life cycle of the operation.

The net present cost (NPC[50]) and the unit reference value (URV[50]) were calculated for the various management options. (Section 2.6.3.1) The life cycle usually considered for engineering projects is 20 years, but for the purpose of this study, the operational life that was used was 50 years[51] with a base date of November 2010.

The capital costs that were considered allowed for the development of a new pumping station, water treatment plants and underground tunnels. The operational costs were allowed for pumping costs, water treatment costs, waste disposal and maintenance of infra-structure. The operational costs are influenced by the specific option chosen, as simulations indicated that the inflow rates would reduce as the basins are flooded. It was assumed that all capital will be borrowed and repaid over a period of 20 years. The consumer price index (CPI) was ranged between 8%, 10% and 12% and the cost of capital (i.e. the effective lending rate) was conservatively assumed at 15%. The 10% discount rate was used for comparison purposes. A trade-off (Section 2.6.3.3) analysis based on the NPC[52] and URVs indicated that (Table 6-8):

1. The costs for Option 4.0 are not applicable as it is not possible to determine the potential costs of downstream environmental impacts and ongoing litigation. The capital costs are zero.
2. The option with the highest NPC of R 4 755 million is Option 4.1 if the operational phase would continue. This is due to the fact that the pumping head is high at 680 m to keep the water level below 900 mamsl. The URV of R 15.65 /m^3 discounted over 50 years. This would also mean that mining could produce the capital for dewatering.
3. Option 4.2 has a NPC of R 2 796 with a URV of R15.64 /m^3, which is similar to the previous option. This is due to the fact that while pumping costs are lower, there is also a directly smaller volume produced which increases the unit cost.
4. Option 4.3 has a NPC of R 2 634 million with a URV of R18.42 /m^3.
5. Option 4.4 has a NPC of R 3 653 million with a URV of R18.58 /m^3.

The option with the lowest capital requirement of R 379 million is Option 4.2, followed by Option 4.3 with R 434 million, then Option 4.4 with R 605 million and the most capital is required by Option 4.1 at R645 million. The capital is driven by the flow rates that determined

[50] The NPC is the NPV in cost terms. The URV is a special case of the NPC calculated with discounted flow volumes over time.
[51] The time frame of 50 years that was used, is the same that is used for other water resources, such as dams (AGES, 2005a).

the size of the water treatment plant, which is the biggest capital component.

If the costs of R2 635 million over a 50 year post-operational period from Option 4.3 is chosen as the long-term sustainable option[53] for the FERB and compared with the income generated of R3 234 billion, over the 126 years of operations, then it represents 0.08% of the income generated. If the minimum average profit is assumed to be 5%, the NPC represents 1.6% of profit (Table 6-7). Had these liabilities been foreseen, it is evident that it would have been possible to make financial provision for the post-operational management and mitigation of the water and environmental problem during the operational phase, when the mines were generating cash. It is a much bigger problem now, since the profits were distributed to shareholders of which most are overseas entities. The problem for the state or any operating company is to obtain the money from cash reserves, which do not currently exist.

Table 6-7 FERB revenue, profit and environmental cost comparison.

Component	Quantity
FERB LoM revenue over 126 years (R mil)	R 3 233 902
Cost of 50 years post-op water pumping + treatment (NPC)	R 2 634
Costs over 50 years as % of revenue	0.08%
Minimum profit margin assumed	5%
Minimum profit (R mil)	R 161 695
Costs (NPC) over 50 years as % of minimum profit	1.6%

In the financial assessment, it was assumed that capital will have to be borrowed to mitigate the water problem (Table 6-8). The assessment indicates that the unit cost[54] ranges from R15.65/ m^3 for Options 4.1 and 4.2 to R18.58 m^3 for Option 4.4. The current price that Rand Water obtains for water is R8.58 m^3 (Rand Water www.randwater.co.za, 12 November 2010). This indicates that the problem cannot be financially mitigated on its own. The only way that any of these solutions are viable, is if the residents of the Greater Johannesburg and Gauteng Province all contribute to the payment of the deficit of between R10 million/month for Option 4.2 to R16.5 million/month for Option 4.4. If Option 4.1 is chosen, then the operating mining company will have to cover this deficit. The unit cost subsidy would amount to between 0.31 cents/m^3 for Option 4.2 to 0.59 cents/m^3 for Option 4.4. The average increase in the water bill for residents would be 0.03% (Table 6-8). This is the only financially viable option.

[52] The NPC for a 15% discount rate and a 10% CPI was used for the assessments.
[53] This option has a natural decant tunnel and does not require pumping, hence the definition of sustainability.
[54] A unit cost compares the capital, finance and operational costs over the life of the project and can be used to make comparisons

All the options were compared and scored in a decision-matrix (Section 2.10), that compares the sustainability (Section 2.5) of the options (Table 6-9). It indicated that Option 4.0 would be socially and politically unacceptable and hence have the highest negative score of 6.4. Options 4.1 and 4.2 have lower scores of 3.8 and 3.4 respectively. The negative aspects are economics for Option 4.1 with the highest capital, NPC, social political and legal aspects for Option 4.2. Options 4.3 and 4.4 are the favourable options in terms of sustainability with Option 4.4 the most sustainable with the lowest score of 2.8. Any option where pumping is required beyond 20 years is not considered as a sustainable solution. Options might be combined e.g. Option 4.2 would be good for a period of 5 to 10 years or maximum 20 years as it is the lowest capital option. It can also be used to test the water flow and quality variation by keeping the water level in the basin at different levels to optimise the final level required. The long-term sustainability plan should then make provision to implement Option 4.4 after a maximum period of e.g. 20 years.

Based on the financial viability and trade-off analysis, it will not be possible to economically manage and mitigate the contaminated water problem in the FERB and the CRB on its own. If the problem is diluted using the water income of the Rand Water distribution network and clients, then the costs of the problem can from technical and financial considerations be diluted to insignificance. It does put a burden on future generations, but the burden is insignificant. This is however not necessarily socially or legally acceptable.

6.2.14.2 Scenario 4b: The potential effects of social (legal and political) aspects

The main reason why the mine water environmental problem was allowed to develop was that historically, there was no understanding of environmental systems, liability and sustainability. The focus was purely on business development and economics without consideration of burdens on future generations. There was also no environmental regulation or legal processes to manage the impacts on the environment.

Although legislation[55] is in place since 1998 to apportion liabilities and keep directors responsible, it would mean drawn out litigation and prosecution of mining companies that does not exist today or that does not have the financial means to cover the liabilities that were created,

for decision-making.

55 National Water Act (NWA) (Act 36 of 1998) and the National Environmental Management Act (NEMA, Act 107 of 1998. Under these acts, companies and directors can be held responsible for environmental liabilities now and in the past.

even if found guilty.

The state has learned from this experience and the new Mineral and Petroleum Resources Development Act (MPRDA, Act 28 of 2002[56]) forces new and existing mines to make ongoing provision for financial liabilities. The provision is either ceded to the Department of Mineral Resources (DMR) or to an independent trust fund.

The present situation is that the DMR, DWA and other state departments were aware of the potential rising problem since 1999. While some of the problems were partially characterised, there was no political will (Section 2.8.2) and hence no decisions were taken to start planning or addressing the problem. The lack of decision-making can be ascribed to the operational dysfunction of the state departments as well as the inability to function cooperatively. This is further exacerbated by the lack of skilled people in these departments that can make the decisions. The political situation can be characterised as an organised anarchy, which acts as positive feedback to the problem and enhances it (Section 2.8.2).

Four sub-scenarios (Scenario 4b.1 to 4b.4) were identified which ranges from the current situation, where Scenario 4b.1 represents the lack of political will to act coherently which leads to business inactivity on the problem and eventual positive feedback that leads to a growing problem with time (Table 6-10, Figure 6-26). In between is Scenario 4b.2 that represents political will but not business buy in. This leads to legal action with implementation of solutions over a longer time period.

Scenario 4b.3 represents a case where business is proactive with the result that no legal action is required. The result is that management and mitigation measures are implemented in shorter time frames. In Scenario 4b.4, it is considered that the gold price soars in the future and the mines are reopened and dewatered. Capital is provided under new legislation to fund sustainable rehabilitation and water treatment. Scenarios 4b.3 and 4b.4 would be the ideal cases to manage and mitigate the contaminated water problem while Scenario 4b.1, which is the current status is the second worst option if the do nothing scenario (Table 6-9) is considered as the *worst case*.

[56] Amended act: Mineral and Petroleum Resources Development Act (No 49 of 2008).

Table 6-8 Scenario 4a: FERB financial assessment on identified management options.

No	4.0	4.1	4.2	4.3	4.4	
Scenario	**Do nothing**	**Operational continue**	**Post-operational pumping**	**Post-operational natural decant via FERB tunnel**	**Post-operational natural decant via FERB & CRB link tunnel with decant tunnel**	**References**
Period (y)	50	50	50	50	50	A period of 50 years were used as water projects such as dams uses these periods.
Flow rate (m^3/d)	50 000	85 000	50 000	40 000	55 000	Measured for Scenario 4.1 and simulated for the other scenarios
Water quality (TDS in mg/L)	2 000	3 000	2 500	2 000	2 000	Conservative assumptions based on measured values
Average mass abstracted (ton/d)	NA	255	125	80	110	Calculated
Cost of capital (%)	NA	15%	15%	15%	15%	Conservative assumption (CPI of 7% + 8% risk estimate)
Capex repayment period (y)	NA	20	20	20	20	Assumed based on capital financing terms
Pump station capex (R mil)	R 0.00	R 50.00	R 29.41			Conservatively estimated
Pumping costs (R/m^3)	R 0.00	R 1.50	R 1.10	R 0.00	R 0.00	Calculated by mechanical engineer based on flow rate and 680 m head based on this month's electricity costs
Pump station maintenance (% of capex)	NA	5%	5%	5%	5%	Estimated
Water treatment plant capex (R mil)	R 0.00	R 595.00	R 350.00	R 280.00	R 385.00	Calculated per m3 based on Emalahleni Water Treatment Works (R250 mil for 30 ML/d). Costs were added to be conservative
Water treatment costs (R/m^3)	R 0.00	R 5.00	R 5.00	R 5.00	R 5.00	WUCs water project high end treatment costs compiled by Prof Maree
Waste disposal costs (R/ton)	NA	R 5.00	R 5.00	R 5.00	R 5.00	Estimated
Water treatment plant maintenance (% of capex)	NA	5%	5%	5%	5%	Estimated
Development of underground decant tunnel FERB (22 km) (R mil)	NA			R 154.00		Calculated from information provided by mining engineer for 1 m wide tunnel.
Development of underground tunnel between FERB & CRB (5 km) (R mil)	NA				R 45.00	Calculated from information provided by mining engineer for 1 m wide tunnel.
Development of underground tunnel from ERPM Shaft to surface 25 km (R mil)	NA				R 175.00	Calculated from information provided by mining engineer for 1 m wide tunnel.
Maintenance on tunnels (% of capex)	NA			2%	2%	Estimated
Total capex R (mil)	NA	R 645.00	R 379.41	R 434.00	R 605.00	Calculated
Nominal costs for first 5 years (R/m^3)	NA	R 16.21	R 16.21	R 19.17	R 19.34	Calculated
Discount rate 1 (%)	NA	8%	8%	8%	8%	Assumption
Discount rate 2 (%)	NA	10%	10%	10%	10%	Assumption
Discount rate 3 (%)	NA	12%	12%	12%	12%	Assumption
NPC 1 (R mil)	NA	R 5 811	R 3 418	R 3 210	R 4 452	Calculated
NPC 2 (R mil)	NA	R 4 755	R 2 796	R 2 634	R 3 653	Calculated
NPC 3 (R mil)	NA	R 4 007	R 2 357	R 2 225	R 3 085	Calculated
URV 1 (R/m^3)	**NA**	**R 15.48**	**R 15.48**	**R 18.17**	**R 18.33**	Calculated
URV 2 (R/m^3)	**NA**	**R 15.65**	**R 15.64**	**R 18.42**	**R 18.58**	Calculated
URV 3 (R/m^3)	**NA**	**R 15.76**	**R 15.75**	**R 18.59**	**R 18.75**	Calculated
Rand water income potential (R/m^3)	NA	R 8.58	R 8.58	R 8.58	R 8.58	Rand Water
Rand water present volumes pumped to Gauteng (ML/d)	NA	3 400	3 400	3 400	3 400	Rand Water
Subsidy required (R mil/month)	NA	R 18.0	R 10.6	R 11.8	R 16.5	Calculated
Water unit cost subsidy (R/m^3)	NA	R 0.005	R 0.003	R 0.003	R 0.005	Calculated
Water unit cost subsidy of current price (%)		0.03%	0.03%	0.03%	0.03%	Calculated

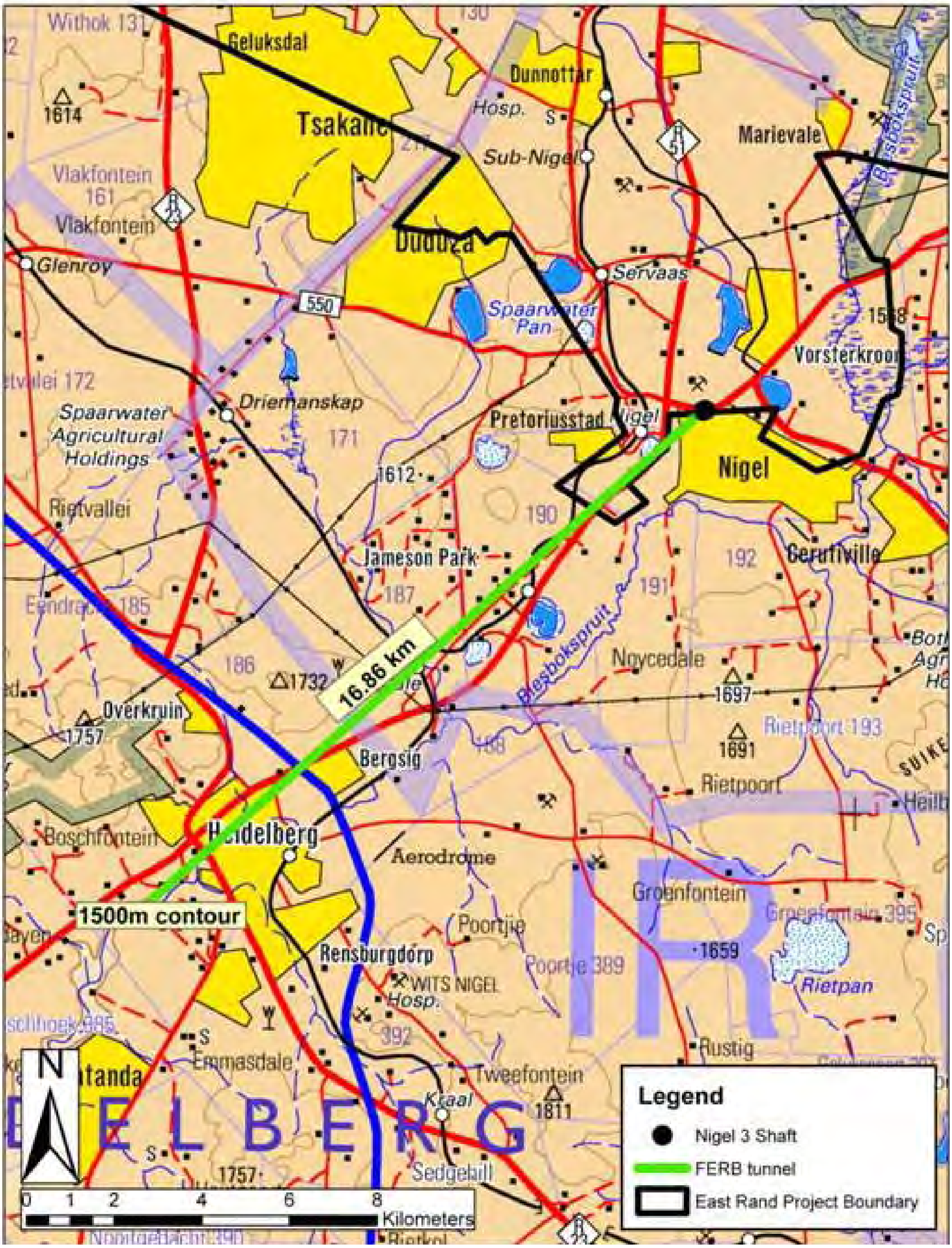


Figure 6-24 Scenario 4a: Option 4.3: FERB post-operational decant tunnel at BCL (1500 mamsl).

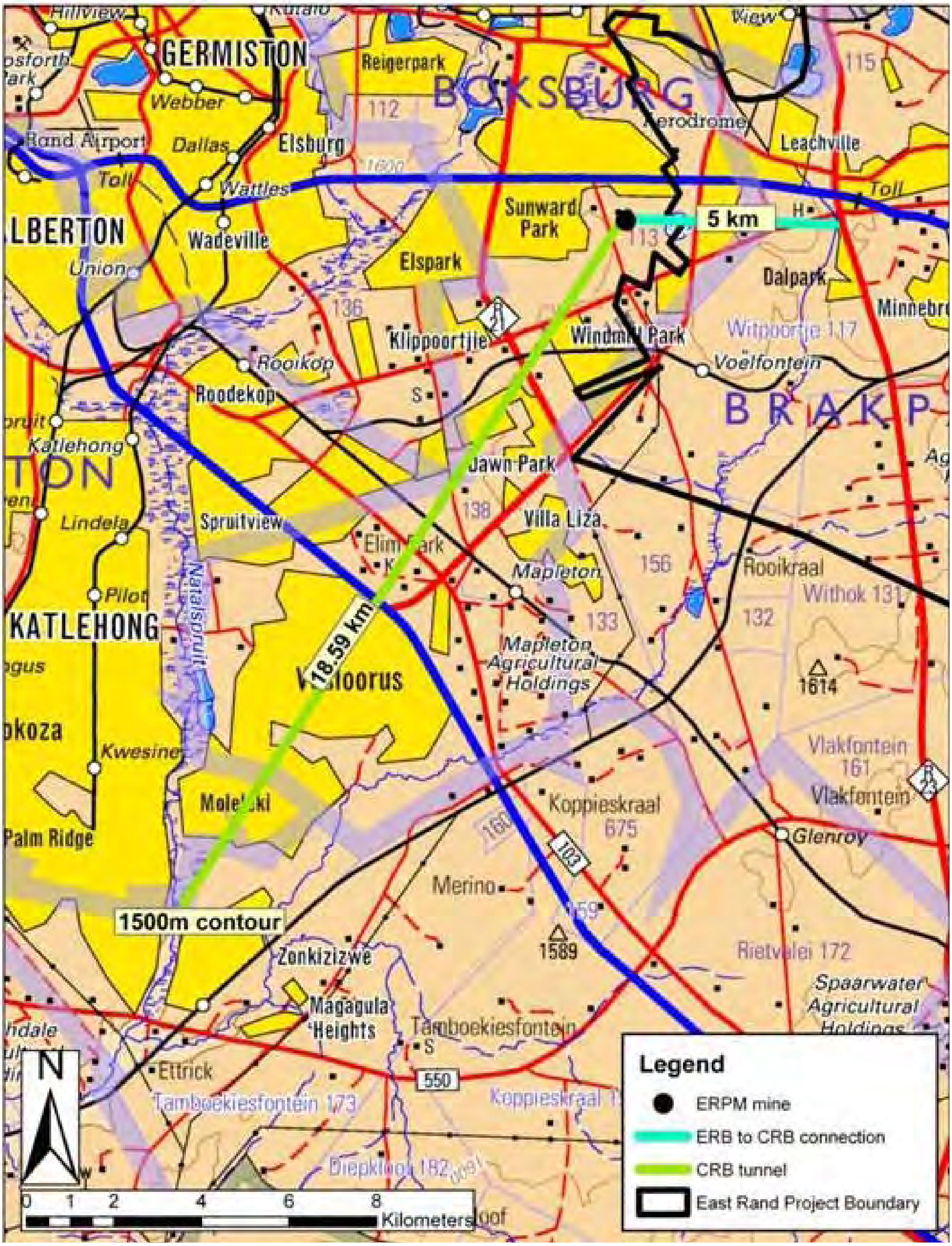


Figure 6-25 Scenario 4a: Option 4.3: FERB & CRB link post-operational decant tunnels at BCL (1500 mamsl).

Table 6-9 Scenario 4a: FERB decision-matrix of management options against sustainability components.

Decision-matrix					
Scenario	**4.0**	**4.1**	**4.2**	**4.3**	**4.4**
Aspect	**Do nothing**	**Operational continue**	**Post-operational pumping**	**Post-operational natural decant via FERB tunnel**	**Post-operational natural decant via FERB & CRB link tunnel with decant tunnel**
Environmental	10	4	4	3	1
Technical	1	4	4	5	5
Business (economics) capital cost	1	5	3	3	4
Social (legal)	10	3	3	2	2
Social (political)	10	3	3	2	2
Total	**32**	**19**	**17**	**15**	**14**
Average	**6.4**	**3.8**	**3.4**	**3.0**	**2.8**
Legend					
Unacceptable	10				
Very high impact	5				
High impact	4				
Medium impact	3				
Low impact	2				
No impact	1				

Table 6-10 Scenario 4b.1: Comparison of political and business sub-scenarios and sub-effects (Figure 6-26, Figure 6-27).

No	Scenario	Effect 1	Sub-effect 2	Sub-effect 3	Sub-effect 4	Sub-effect 5
1	Scenario 4.b.1: Post-operational Phase with lack of political will	Business is not activated to act and spent capital on the environmental problem	Legal action is not taken and state intervention is slow and incoherent	The ingress and decanting problem escalates (positive feedback) due to erosion of mine residue facilities and ponding of surface water. Neighbouring social groups start to apply pressure, especially using the media	Action is taken but no holistic plan is in place as there is not sufficient time for research & development.	Implementation of the best available mitigation measures such as treatment is done as opposed to better methods that could have been developed through ongoing research and development
2	Scenario 4.b.2: Post-operational Phase with political will	Business is not activated to act and spent capital on the environmental problem	Legal action is taken and state intervention is quick and coherent	Mine residue facilities are rehabilitated, surface water management measures implemented and water treatment plants are build (negative feedback) to manage the flow and water quality within acceptable standards. It however takes place after a protracted litigation period	The environmental water problem is mitigated and managed to become a water supply solution	
3	Scenario 4.b.3: Post-operational Phase with political will and private sector buy in	Business is pro-active to act and spent capital on the environmental problem	Legal action is not required. Only regulatory guidance	Mine residue facilities are rehabilitated and water treatment plants are build to manage the flow and water quality within acceptable standards (negative feedback). The time frames are relatively short as there is pro-active support from business	The environmental water problem is mitigated and managed to become a water supply solution	
4	Scenario 4.b.4: Future operational Phase with financial gains and remining due to high gold price	Business is pro-active to act and spent capital on the environmental problem. The mines are dewatered again for operational purposes.	Legal action is not required. Only regulatory guidance	New mine residue facilities are designed to minimise environmental impacts. Surface water management measures are developed (negative feedback).	The environmental water problem is mitigated and managed to become to remain a water supply solution	

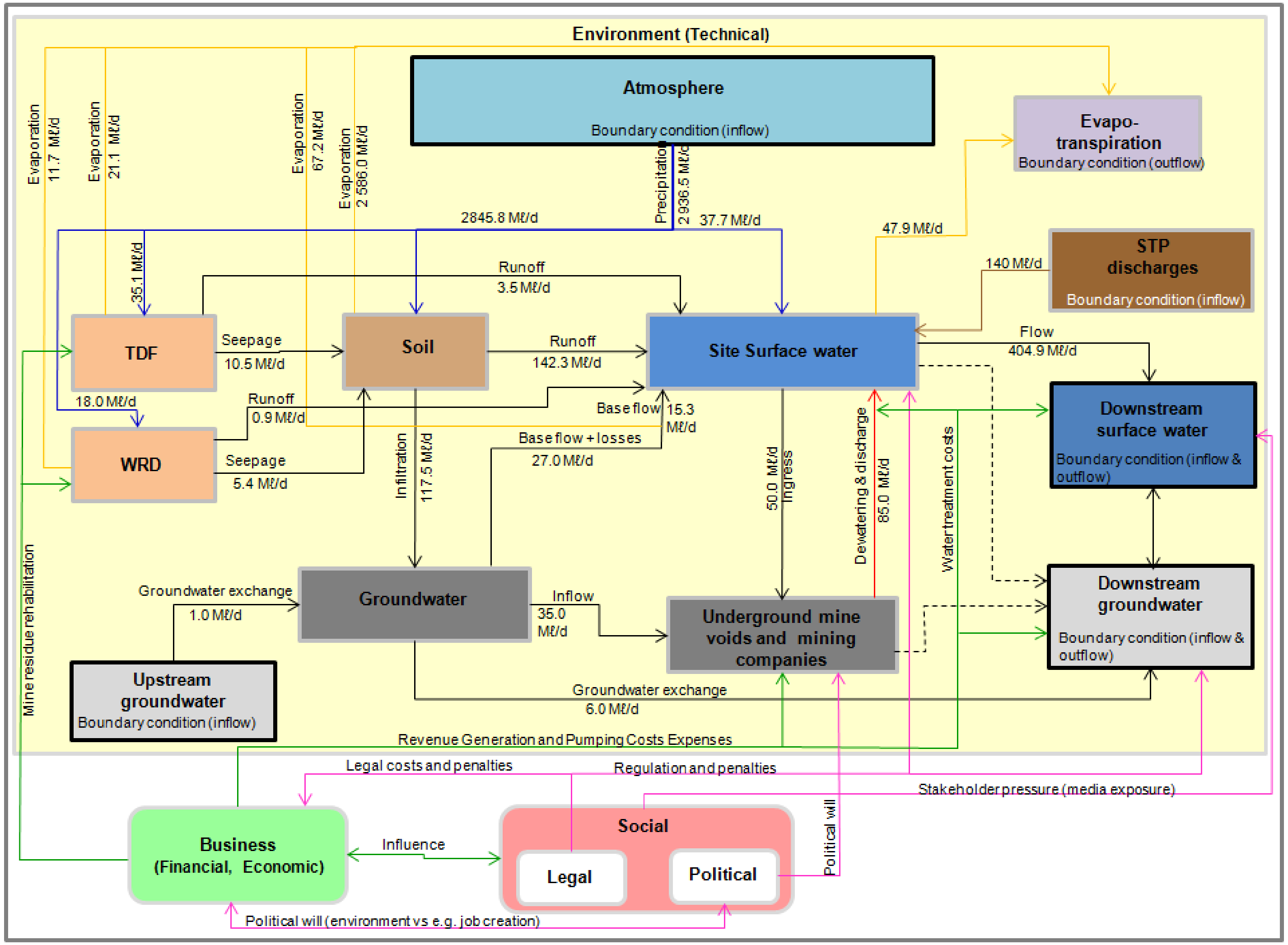


Figure 6-26 Scenario 4b.1: Process flow model for the post-operational phase with do nothing and no political will (Table 6-10).

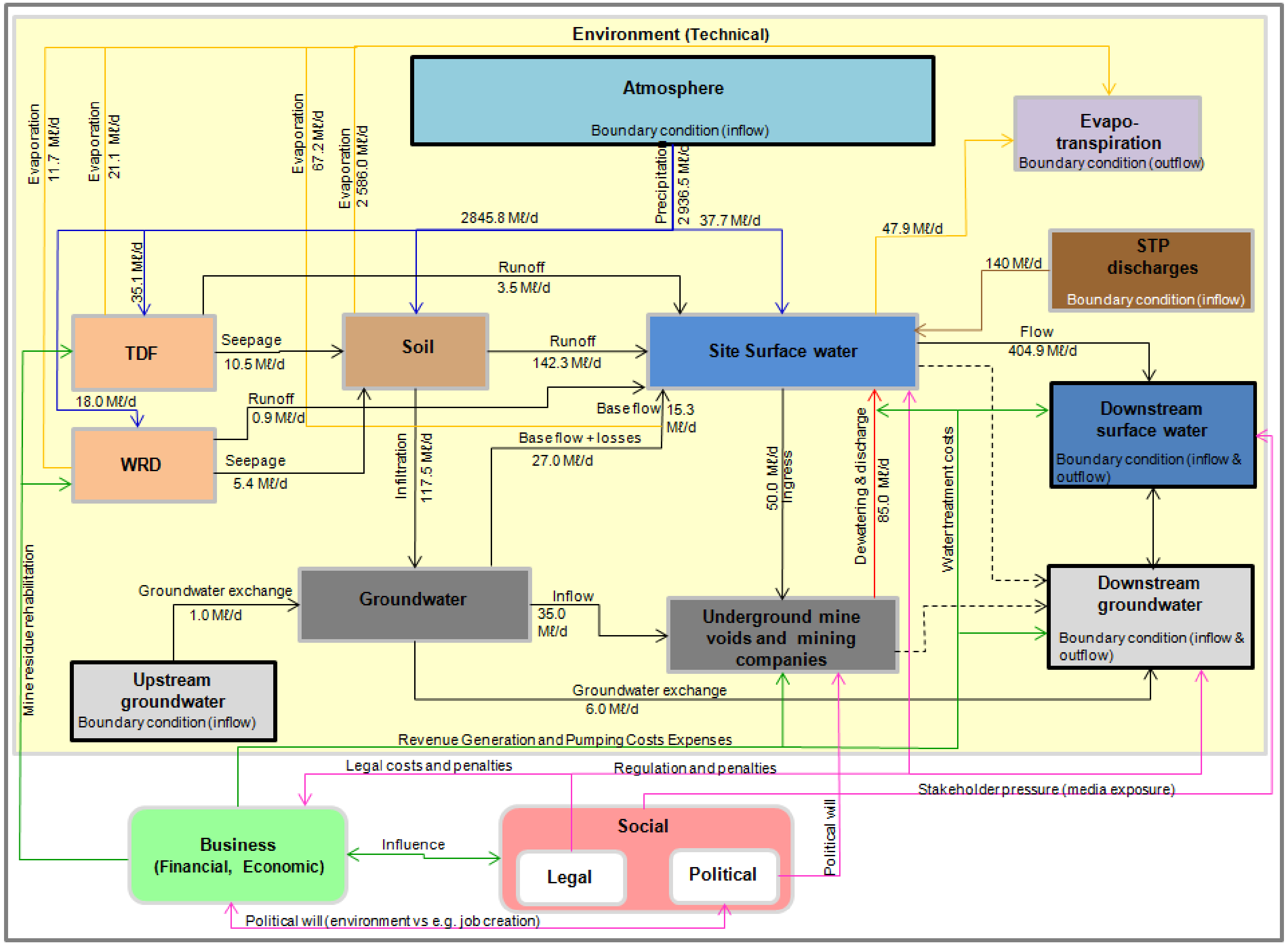


Figure 6-27 Scenario 4b.3: Process flow model for the post-operational phase with political will and management measures (Table 6-10).

Except for Scenario 4b.1, if managed correctly the environmental problem could become a water supply solution for the Greater Gauteng Area. The question is whether it is right to let the current generation pay for a legacy left by the previous generation? One way of viewing this, is that it is not fair to do that. This will lead to the state having to provide the finance. The other way to view this is to accept that the financial and developmental gains that developed Johannesburg and the Greater Gauteng Area in the last century as the leading capital in Africa is directly due to gold mining and that an increase of 0.03% in the water bill of every resident is not too much to ask. In the one instance, all the residents of the Gauteng Area will pay for the mitigation measures. If this is not done, the state will have to provide the finances anyway, which would mean that all the taxpayers of the whole country will have to pay for the mitigation measures. Both of these options may then be viewed as equally wrong, but necessary to prevent future adverse environmental impacts.

6.3 Evaluation of results

The evaluation of results must put the output in perspective in terms of the purposes of decision-making. Now that the problem boundaries, components, interactions and scenarios are determined, it becomes like a chess game (Section 2.9.6) where the process should be optimised (Section 2.9.7) to determine what the primary and higher order effects of the environmental decisions could be. This was already done in the previous phases with the identification of management Options 4.1 for the first 10 to 20 years followed by the implementation of Option 4.4 for the long-term solution. It will be important to flood the basin to minimise ingress and the presence of oxygen that is mainly responsible for acid formation.

An important aspect of a systems model is that it is usually controlled by one constraint (Section 2.6.2; Goldratt, 1990). The single most important constraint is the availability of capital (economics) supported by the historical lack of political will to manage and mitigate the problem by instituting legislation (Mining Weekly, 2010b). The most important technical constraints are the lack of rehabilitation of mine residue facilities such as tailings dams and waste rock dumps and the availability of advanced water treatment technologies that can be ascribed to a lack of research and development that is due to a historical lack of political will. The problem can be reduced significantly from a technical perspective if mine residue facilities are rehabilitated and surface water ponding minimised. Should the civil structures of the old roads be removed and culverts enlarged below existing municipal roads, together with rehabilitation of mine residue facilities, the problem can be reduced significantly. The simulations indicated that this could reduce the problem by up to 20% (from 85 Mℓ/d to 68 Mℓ/d) and it can be done at relatively low cost and it is not technically difficult to achieve.

6.3.1 Scenarios and uncertainty

It is important to note that the scenarios that were developed and simulated may not represent the actual field case and that not all the information was available before the initiation of the systems models. The purpose was also not to determine the actual field case, as this is accepted to be an impossible task. From all the data that is available, only the mine dewatering rate, rainfall and water levels could be measured. Based on this data, the regional water flow and groundwater flow balances were simulated. One of the outcomes was that approximately 50 Mℓ/d of the 85 Mℓ/d originate from surface water ingress, while 35 Mℓ/d originate from the regional groundwater (Malmani Dolomite and Karoo Aquifers) (Figure 6-13). This finding was unexpected as the expectation was that most of the groundwater would originate from the dolomite aquifers. The questions are:

- Could the water balances be different and what influence would it have on the decision-making process?
- How do we consider uncertainties in such a complex problem?

A process was developed to determine whether these scenarios were firstly possible after which probable scenarios were determined.

6.3.2 Critical evaluation of scenarios based on the judicial decision-making methodology

If the analyst would be cross-examined about the validity of the model, there is no definitive way to defend it, as there could be a number of water balance versions that could be arrived at using the simulations. Since only three variables namely; rainfall, the dewatering rate and the water level elevations in the basin and in the surrounding aquifers are known, the water balances could very well be different. The process followed accepts that the actual case is unknown and aims to determine the problem and answer boundaries rather than to arrive at a specific answers. This is done by following a process of elimination and cross-examination. The cross-examination can also be considered as inverse testing of scenarios by following a process that would be found in judicial processes (Section 2.7). Inverse testing is when a model is used to determine a minimum or maximum value to explain a measured value. It is done by e.g. asking what the minimum recharge should be to explain the measured dewatering rate and then to test whether this is possible or not.

Below are a number of critical questions asked internally as part of the cross-examination process to evaluate the validity of results. It is an internal cross-examination of the scenario results to determine whether it would stand up to scrutiny and to build a case for each scenario.

The first question is what can be termed a boundary question and evaluates extreme ends. It is analogue to the binary approach (0/1) (Section 2.7.3) and seeks to understand what is possible and what is impossible before it aims to understand what is probable.

How do we know that this water balance is correct?

The water balance is important as the management decision-making is based on it. The system is so complex and data is so scarce with uncertainty so high, that it is not possible to determine the *actual* water balance as it is in the field. It must be accepted that to determine the actual case is not possible. To aim to achieve this will require years of scientific measurements and research. While research and measurements are important in providing information in the decision-making process, the constraint is time and cost. If an analyst seeks to obtain the perfect information, it would lead to a situation of *doing research while the house is burning down*[57]. There are also variable factors e.g. new sink holes that form, which could enhance ingress and change the situation on an ongoing basis. The water balance is therefore not *correct* as it is in the field. This was not the aim of developing the water balance. It is here where a distinction should be made between what is *scientifically correct* and what is *sufficient for decision-making*. In the decision-making process, an iterative process is followed by first using existing data to provide the information for a first iteration of decision-making. This is where the difference between having an objective to reach a decision and aiming to collect data and information is important.

The collection of any data should be to provide information for decision-making, otherwise it could be a waste of time and money. In terms of data that were collected or measurements that were made, it is important to distinguish between the components that (Figure 6-26, Section

[57] This is also known as the analysis paralysis problem that is often found when some academics engage in decision-making.

4.2.1):

1. Can be measured and that have a low degree of uncertainty. These are the mine dewatering volumes, rainfall, groundwater levels, STP discharges and surface water flow.
2. That must be calculated based on the measured components and have a high degree of uncertainty. These are surface water runoff volumes, and evaporation volumes.
3. Where no data exists and assumptions or qualified estimates have to be made. These parameters are; groundwater recharge, mine void ingress, aquifer storativity and porosity, business and social aspects.

It was shown that business and socio-political aspects are the most important controlling factors that lead to the creation of the problem (Section 6.2.14). These aspects are not measurable, but can be evaluated in a systems process to evaluate scenarios for the purposes of decision-making. It would therefore be difficult or even impossible to defend the water balance in court, but it will be possible to defend the decision-making process as will be discussed later in this chapter.

Can only the groundwater recharge in the catchment explain the dewatered volume?

The total groundwater recharge across the regional catchment is in the order of 117 Mℓ/d. It can explain the dewatered volume without inflow from outside the catchment area. However, if the dewatered volume of 85 Mℓ/d originated only from groundwater, it would have depleted almost 73% of the aquifer over time. One would expect the aquifer to be largely dewatered. The water level data indicates that the average depth to groundwater is 16 m, which indicates that the regional aquifers are mostly saturated (AGES, 2005b). Geological data from a number of mine exploration boreholes shows that there is a regional and thick diabase sill known as the Green Sill that lies laterally across the area (Figure 6-2, AGES, 2006). This sill acts as a barrier to vertical groundwater flow. Mining mostly takes place at depths far below the regional dolomite aquifer. Inflow of groundwater takes place via shafts and shallow mined out areas but it is highly improbable that groundwater can explain the total dewatered volume.

Can the surface water runoff in the catchment explain the dewatered volume?

The surface water runoff including STP discharged is in the order of 288 Mℓ/d. If it were the case that both surface water runoff and groundwater recharge would have been less than the mine dewatering volume of 85 Mℓ/d, then it would be important to determine that the problem boundary is not correct and that water from outside the boundaries is required to explain the dewatered volume. In this case, the groundwater recharge, surface water runoff and STP discharges totals 405 Mℓ/d (Figure 6-26), which is almost 5 times the dewatering volume. Only the surface water and STP discharges of 288 Mℓ/d can explain the dewatered volume by 3.4 times. Permeability tests done along Cowles Dam, located 2 km north from Grootvlei No 3 Shaft (Figure 6-11) indicated that it could be responsible for the total 85 Mℓ/d. The head gradients measured in boreholes along this dam verified that there was a gradient from the dam towards the subsurface, but it was not steep enough to leak at the full dewatered volume because of the lower permeability of the underlying bedrock (AGES, 2005b).

It is therefore possible and probable for surface water ingress to explain the dewatered volume.

Are there other sources that could contribute to ingress into the mine voids?

Apart from the surface water runoff, STP discharges, which forms part of the surface water and groundwater components, there are no other sources. These sources are sufficient to explain the problem and for the identification of management decisions. The detailed numerical groundwater modelling was used to scale the probabilities of surface water and groundwater ingress to a more accurate value.

Is it possible to determine how much water originates from groundwater and how much from surface water?

Now that it was determined what is possible and what is impossible, probabilities on the sources can be evaluated. This is an indirect process for which calculations such as the regional water balance (Figure 6-13) or more detailed groundwater balances are used (Table 6-3, Figure 6-8, Figure 6-7). In this evaluation, the components representing the mine void and groundwater were evaluated in more detail by using numerical groundwater simulations (Section 2.9.8.3 and Section 5.6). These models assist the decision-maker to obtain more detailed information by using indirect methods that calculate the outcomes of scenarios. It indicated that around 70% (60 Mℓ/d) of the ingress is probable from surface water and 30% (25 Mℓ/d) from groundwater.

Did we have enough data?

The data that were used were (Table 6-1, AGES, 2006):

1. Mine dewatering flow records obtained from Grootvlei Mine.
2. Two surface water flow gauges installed by the Council for GeoScience (CGS).
3. Geology spatial data from the CGS.
4. Water level data from 58 boreholes distributed across the study area (Figure 6-4).
5. Pump tests from 8 boreholes.
6. Rainfall data obtained from the SA Weather Bureau.

The adequacy of data depends on whether it is adequate for decision-making. In this case, the data was *sufficient* to develop a regional water balance and to perform simulations on the mine void and groundwater system. Data and parameters that were not available or even undeterminable such as the mine void volume and porosity of the various hydrogeological layers were based on qualified estimates and assumptions.

The data that is available is not optimal, but is considered *sufficient* for a first level of decision-making but based on the decision-making process defined iterations should be done once the models are used to determine which parameters are important for the decision-making process (Section 3.3, Section 5.6, Figure 5-8).

What if the assumptions are wrong? Or too many assumptions are made?

In the absence of perfect data, assumptions have to be made (Section 3.7, Figure 3-27). If assumptions are not made, then it is not possible to make decisions. In terms of assumptions, it is important how these assumptions are made. The way in which important assumptions such as the porosity of the mine void was made, which controls the rewatering rate is important (Section 6.2.13.3). The lowest porosity of Main Reef of 10% was chosen to base management decisions on as it provided the quickest rewatering rate of 5 years (Table 6-6). The higher porosity of the Main Reef at 30% was assumed, then the decanting is likely to take 24 years. The question that must be asked is: *What is the consequence if my assumption, model or outcome is wrong*? This is the falsifiable approach that was followed by Popper (Section 2.9.2). If the assumption chosen is wrong, then the decanting would take longer with the result that the problem does arise as a surprise with management and mitigation measures that are not in place timeously.

This is where field tests or monitoring should take place to quantify these important parameters so that another decision-making iteration can be made (Section 5.6, Figure 5-8). The value of the systems approach and modelling in the decision-making process is that important parameters are now identified that could focus data collection.

Which scenario has the best and the weakest case?

Based on the judicial process approach, the present day Scenario 2 has the best case. The proof for this scenario is based on the measured flow rates from Grootvlei No 3 Shaft that dewaters at rate of 85 Mℓ/d. The head elevation in the mine at 900 mamsl is measured, as well as the mine water, surface water and groundwater quality parameters. Additional measurements will have to be made to strengthen the case. These measurements should include the following:

- Spatial water quality distribution.
- The rate of rise if pumps are e.g. shut down for 5 days.
- Surface water runoff, especially during flood conditions.

The futuristic Scenario 4 has the weakest case. Aspects that were not or cannot be measured are; recharge, Main Reef and aquifer porosity, future water quality, water levels with time, decant rates with time and the decant areas. These aspects were calculated using analytical and numerical models. Of the mitigation and management considerations, Option 4.4 has the best case, both from an economical and sustainability perspective (Table 6-9).

6.3.3 Sensitivity analysis

Sensitivity analysis is when the influence of model parameters is used to determine the influence on the model outcomes. It is very useful as it can be used to identify the most important parameters that could be used to guide data collection programmes (Section 4.6). A sensitivity analysis is also done on parameters that cannot be determined in the field, but for which reasonable assumptions can be made in terms of a range. It was done on the porosity of the FERB mine void (Section 6.2.13.3, Table 6-6). One of the main uses of sensitivity analyses is to address uncertainties (6.3.1). It showed that the porosity (i.e. volume) of the Main Reef was the most important parameter controlling the decant rate.

6.3.4 The use of models in inverse testing

Inverse testing[58] is where a model is used in an inverse way to determine what a parameter value must be to provide a known answer. An example would be to determine what must the permeability of the sediments in Cowles Dam be to seep 85 Mℓ/d? Cowles Dam has a surface area of 92.8 ha (AGES, 2005b). Under a piezometric gradient of 1, the minimum permeability required for the sediments to allow for this seepage is 0.09 m/d (Equation 2-4). This is possible, but considered a low probability as the sediment hydraulic conductivity is in the silt fraction range of 0.001 m/d to 0.008 m/d. Inverse testing can in this way be used to ask the reciprocate question to determine what minimum or maximum parameter values should be, to explain a measured (i.e. known) value.

[58] Note that this is not inverse modelling.

CHAPTER 7

"Simplicity is the ultimate sophistication" Leonardo da Vinci

7 DEVELOPMENT OF AN ASSURED METHOD FOR DECISION-MAKING IN COMPLEX ENVIRONMENTAL WATER MANAGEMENT PROBLEMS

7.1 Introduction

The application of systems thinking in decision-making for a complex environmental water management problem such as the FERB mine flooding and decanting, indicated that it is possible to arrive at outcomes that are sufficient for decision-making. The application of decision-making criteria on the case study are defined further in this section within the assured method for decision-making. This method is based on the decision-making principles determine from this study and aims to enable decision-makers to reach management decisions with data that is often sparse and associated with a high degree of uncertainty.

7.2 The assured systems model method for environmental decision-making

One of the aims of this study was to determine how decisions can be made when data is sparse, unavailable or even unknowable (e.g. recharge, porosity, storativity etc). The concern was addressed that a pure scientific approach that is applied superficially would require *a lot of* or *exact* data prior to making any decisions. It was shown that science does make use of hypotheses (Section 2.9.2) and assumptions (Section 3.7). The difference between the statistical Bayesian and Frequentist approaches is an example of different approaches to arrive at conclusions within the scientific community. The Bayesian approach is often favoured as it allows decision-making using sparse data, within an iterative approach in which the answer converges to within an acceptable error (Section 2.9.4).

In environmental management, the influence of subjective aspects, such as business and politics must also be accounted for. Decisions have to be made within the constraints of data scarcity, uncertainty, time and budget. The Bounded Rationality Model for decision-making is favoured as a practical method that can be used to arrive at answers using *sufficient* information (Section 2.6.2). The Rational Model for decision-making requires a more analytical and detailed process that requires *optimal* information before decisions are made (Section 2.6.1). The problem with this method is that is generally too detailed and takes too long in terms of time and money (Section 2.6.2). The Bounded Rationality Model was found to be more effective and used more often by decision-makers in the business environment.

The principles of sustainability should be considered in environmental decision-making, which means an integration between scientific (technical), business (economics) and social (legal and political) spheres (Section 2.5, Figure 2-3). Based on these principles, a decision-making method is developed that is based on an *assured approach*. The assured method for decision-making, is based on the principle of *falsifiability*[59] and the *minimax*[60] rule. Based on the principle of falsifiability, the question is asked; *what will happen if the assumption or model or method outcome is wrong*? It does not only consider what will happen if the assumption is right. This is the method used by Karl Popper to define what science is (Section 2.9.2, http://plato.stanford.edu/entries/popper/#ProDem/). Based on the minimax rule, the question is; *what is the minimum time frame in which rewatering could take place*? Or *what is the maximum volume that is likely to decant*? The minimax and maximin philosophies were developed in game theory to bring certainty to probabilistic game theory (Luger, 2002; Boutilier et al. 2006; Section 2.9.6);) and is chosen to limit or even remove uncertainty from the decision-making process (Figure 7-1; http://library.thinkquest.org/26408/math/minimax.shtml). If the aim is to determine a minimum or a maximum, rather than the actual value or status, it simplifies the quantitative process for the purposes of decision-making.

The assured model for environmental decision-making is defined in a step-wise and iterative decision-making strategy or method , based on the following expansion of the systems modelling process for decision-making (Figure 7-2):

1. Define the problem objectives.
2. Develop a strategy map or model for the problem.
3. Determine the problem boundaries. These are termed boundary scenarios.
4. Based on the strategy map, develop a conceptual model for the problem. Accept that data

[59] Falsifiability or refutability is the logical possibility that an assertion could be shown false by a particular observation or physical experiment. That something is "falsifiable" does not mean it is false; rather, it means that if the statement were false, then its falsehood could be demonstrated (http://plato.stanford.edu/entries/popper/#ProDem/).
[60] Minimax is a strategy of game theory (Section 2.9.6) to minimise a players potential losses while maximizing the potential gains. Maximin is the procedure of choosing the strategy that least benefits the most advantaged member of a group (http://dictionary.reference.com/browse/minimax). Of specific importance, is that the minimax approach is non-probabilistic and is used based on an evaluation of scenarios in the absence of sufficient data.

will be imperfect, in most cases sparse and associated with a high level of uncertainty.

5. Evaluate and interpret existing data using deterministic and statistical methods.
6. Develop a systems model that can serve as the basis of problem qualification or quantification. More detailed analytical or numerical models can be developed for model sub-systems or specific components of interest (Figure 5-2).
7. First analyse existing data, then plan and optimise collection of new data. Make use of data worth techniques.
8. Identify and develop required scenarios within the problem boundaries.
    i. Use the *principle of falsifiability* by asking what is the influence or potential effects if the assumptions are wrong? Do not only consider what could be the outcome if assumptions are right.
    ii. Make use of the *precautionary principle and the minimax rule* when making assumptions for decision-making purposes. This might mean that the assumption is scientifically incorrect, but acceptable for the making of management decisions. If parameters are unknown, e.g. aquifer storativity or number of wetlands that use groundwater in a specific catchment, then conservative or extreme assumptions are made. By assuming that storativity is very small or even zero, the analyst scales the problem beyond the domain of probabilities[61] (Figure 7-1). If the principles of *falsifiability* and *minimax* are applied, then it means that if a mistake was made, then the decision can be defended.
    iii. Identify and evaluate constraints in the systems process flow model. Evaluate measures to manage these constraints.
    iv. Evaluate solutions (i.e. management and mitigation or value creation measures) that could be obtained by a combination or sequence of the scenarios. This is tactical decision-making.
9. Simulate scenarios using qualitative and or quantitative models, depending on the phase of the assessment.

[61] In the case of a mine dewatering problem, the assumption would be that the storativity is very high. The nature of the assumption depends on the nature of the problem evaluated.

10. Critically evaluate results:

    i. First test what is possible and what is impossible before testing what is probable. Identify the most and least defendable scenarios by determining what is possible and impossible (1 or 0) before determining what is probable or improbable (0<1).

    ii. Interrogate input data to distinguish between certain and uncertain data.

    iii. Interrogate and cross-examine the validity and assumptions of scenarios.

    iv. Interrogate the outcome of scenarios based on the judicial decision-making method.

    v. Make use of inverse testing using the models.

    vi. Make use of extreme ends for scenario outcome testing. The testing of extreme ends are to make the assumption that e.g. all water that ingress into the FERB originates from groundwater and to then test whether this is firstly possible or impossible and if it is then to evaluate the probability (Section 3.3.3).

    vii. Develop a decision-matrix that can be used to weigh alternatives (Table 6-9).

    viii. Identify and determine the most probable scenarios for which the best case can be put forward. Follow the process used in judicial processes and build a case for and against each scenario. The scenario/s with the best case is usually the best option/s.

    ix. Perform financial, legal and social evaluation as part of the modelling process.

11. Re-iterate to any of the steps between 2 and 11 above until information is *sufficient* or if required based on the consequence of the decision *optimal* for decision-making.

12. Make decision.

13. Monitoring of decision outcomes with time and adjust or re-iterate in the decision-making process if required.

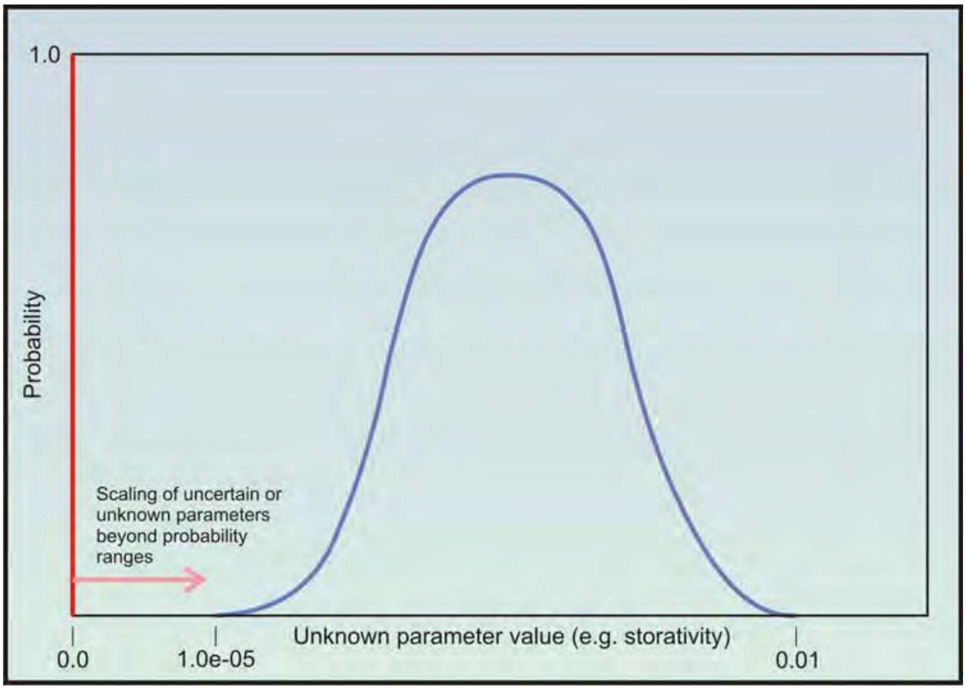


Figure 7-1 Schematic representation of the conservative minimax assumption approach used in the assured model.

The assured model for decision making is a process that could stop at any point and re-iterate within itself. This occurs while the level of understanding of the decision-maker is elevated during each iteration or problem that is encountered in the decision-making process, which forces iterations.

The advantages of this process are:

- It allows decision-making in complex environmental problems using limited and sparse data that is associated with a high degree of uncertainty. In extreme cases, decisions can be made with no direct information. In these cases, analogue information or assumptions can be made.
- If assumptions prove to be too conservative, the approach can be adapted[62].
- It focuses and optimises data collection in each iteration. If the data worth principle is applied, costs could be saved on data collection programmes.

[62] Such as that if e.g. it is assumed that all the wetlands in a certain catchment use groundwater (Information Box 6-A) and the outcome shows that based on the assumption made, it may use more than double the volume that can realistically be recharged from rainfall. Based on the minimax rule and precautionary principle, it can be determined what the maximum groundwater use from wetlands could be in a specific catchment. Ranges could also be applied between an extreme value such as a storativity of zero in water supply problems or the use of a base minimum value to demonstrate the effect of e.g. storativity on the groundwater yield.

- It saves on expenditure as in the case of the Outeniqua Groundwater Study (Information Box 6-A; AGES, 2010b), it indicated that boreholes were not the most important groundwater sink. The purely scientific or reductionist approach was to collect more and more borehole data while the holistic conceptual model that was developed in the assured model iterations indicated that forestry and wetlands could use 10 times the volume of groundwater.
- The approach follows the judicial process analogy where a case is developed for and against each scenario (Section 2.7). It is the best approach to follow when there is a chance that it should to stand up in court.

A basic version of the model that was formally developed here was applied by the author on several complex groundwater problems in the past 5 years (AGES, 2005a, AGES, 2005b, AGES 2006, AGES 2010b). The outcome in all these applications, such as the FERB, was that from a seemingly *pure scientific* perspective, analysts required more and more data and as it became available, more and more uncertainties surfaced. The process formed a *divergent progression* away from decision-making and could ultimately have been responsible for the lack of political will in the case of the FERB. Scientists with varying opinions that did not follow a decision-making process got *lost among the trees* and were focusing on data and uncertainties by arguing about facts and figures while losing sight of the *forest* that generated doubt with political decision-makers.

The purpose of scientific studies in (complex) environmental problems should be for decision-making. If the purpose of these studies does not provide the direction before it commences, then it is like trying to get to a destination by starting to drive. *If the destination is not determined first, then the faster one drives, the quicker you arrive at the wrong place* (Covey, 1989). The purpose of the assured model for environmental decision-making is to first find the destination before the journey is attempted.

Information Box 6-A

In the case of the Outeniqua Groundwater Reserve Study (AGES, 2010b), it was not known which wetlands use groundwater and which not. Wetlands were identified as potentially important groundwater sinks in the system and the project time and budget constraints did not allow for additional fieldwork to be done. Some analysts felt that the project could not and should not continue with these levels of uncertainty.

The problem is that there are numerous unknowns, such as aquifer storativity, recharge etc, which means that on a *seemingly pure scientific basis*, no decisions can be made regarding groundwater component of the reserve (Information Box 3-A) in most areas of the country. This causes the problem to grow while research is being done. Based on the precautionary principle, it was decided to assume that all wetlands use groundwater. This is not scientifically correct, but the project time and budget constraints did not allow for additional field work that could cost millions of rands. The models were run with this assumption and it indicated that 7 out of the 18 sub-catchments are potentially stressed beyond the level that water use licenses for additional groundwater use could be allowed. By using this assumption, the client saved to incur costs to do detailed wetland field studies for 11 of the other catchments. The notion could be that the outcome is unfair towards potential groundwater users. If the context of the assessment is taken into account and the regulatory authority as well as other stakeholders understand the context of the assured model, it does not mean that groundwater in those 7 sub-catchments are really unavailable. This assumption had to be made to allow for a first iteration of decision-making on the groundwater component of the reserve. If a prospective developer wants to make use of groundwater in any of those so called stressed catchments, then additional fieldwork can be done to *prove* e.g. that the wetlands do not use groundwater. That allocation is then unlocked and provided to the developer. The assumption did however allow the regulatory authority to come a decision point and be on the safe side if mistakes were made based on assumptions.

In this case study, the assured model that was applied on the FERB is considered as a first iteration on a strategic level. It is therefore considered to be at a concept level with additional iterations required to collect data on specific important parameters such as the rate of rise of the water level and costs of water treatment technologies. These additional iterations should be elevated to pre-feasibility study (PFS), feasibility study (FS) and detailed design before any implementations can be made.

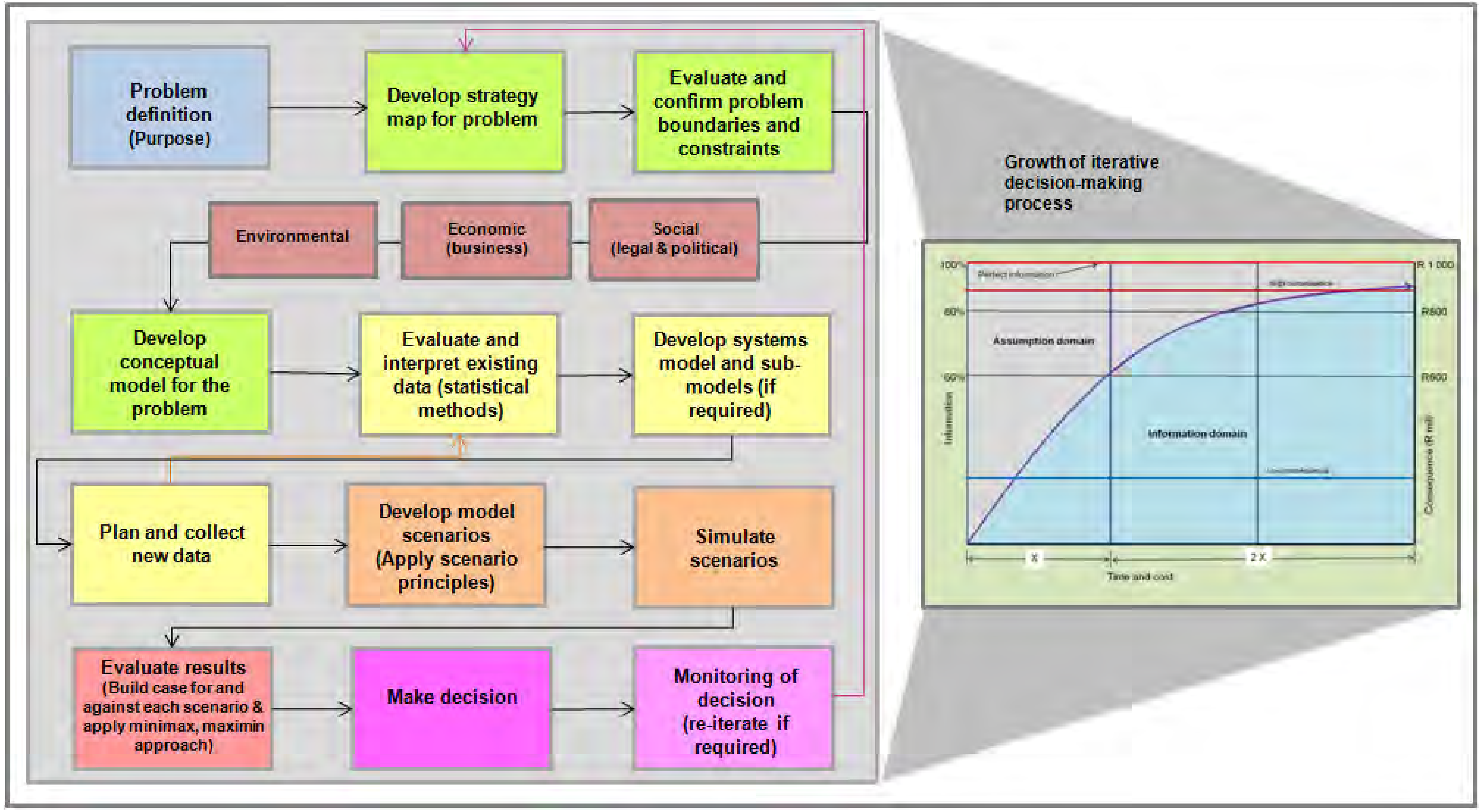


Figure 7-2 Schematic representation of the Assured Model decision-making method.

7.2.1 Application of the assured systems model to the Outeniqua Reserve and Middelburg water supply problems

Criticism on the assured systems model method for environmental decision-making could be that due to the application of the *precautionary, minimax and falsifiability* principles, it could create perception problems. Due to the bottom-up or top-down (i.e. minimax) approaches followed in the *assured method*, it could be perceived as *risk-averse* and providing *not the correct answer*. Considerations of these views were encountered in a number of field studies, such as the Outeniqua Groundwater Component Reserve Study (AGES, 2010b) and the Middelburg Water Supply Study (AGES, 2010c). The critical review in the case of Outeniqua, was that there could be much more groundwater available, in the order of 220 Mm^3/a (Parsons, 2006), than the minimum value of 6 Mm^3/a (AGES, 2010b) that was indicated by following the assured method. In the case of the Outeniqua debate, the principle of falsifiability protects the regulator who must make use of these volumes for the implementation of regulatory limits for the purposes of the Water Reserve (Section 4, Information Box 4-A). If in the Outeniqua case the assured model was wrong, the regulator would not have a legal problem on its hands whereas if the advice was followed as was determined in previous studies and it was wrong, then the regulator could face future legal problems. This is due to the fact that neither the minimum of 6 Mm^3/a, nor the 220 Mm^3/a can be proven. But it would be much easier to defend a minimum of 6 Mm^3/a than an actual volume of groundwater of 220 Mm^3/a, which is impossible to prove.

The difference in output was that the large value of 220 Mm^3/a, was based on a pure scientific approach that seeked to determine the *actual* volume of groundwater available, which is actually impossible and not useful for the purposes of decision-making. The application of the assured method seeked to determine a *minimum* value based on the minimax approach as it accepted that the actual value is unknown. By aiming to determine the actual value, it makes it impossible for regulators to make decisions as any output could easily be contested in court and debated for years as the *actual* answer varies with time and cannot be known. What can be determined is a *minimum* or a *maximum (i.e. minimax or maximin approach)* which can be tracked with time to converge between an envelope (Figure 7-3).

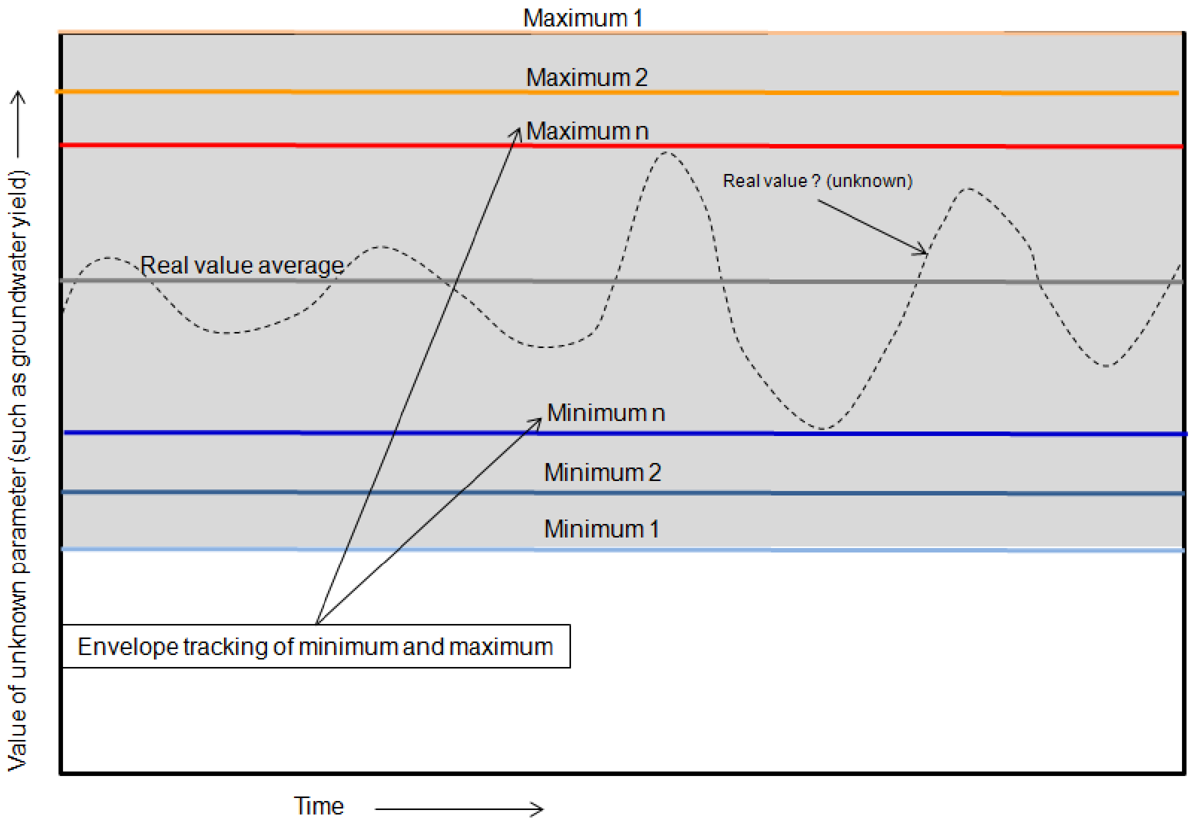


Figure 7-3 Schematic representation of the envelope tracking used in the assured model.

In the Middelburg Water Supply Study (AGES, 2010c), the assured method was used to determine a groundwater yield that is higher than the "sustainable yield" determined by groundwater specialists who followed a pure scientific and risk-averse approach (Annexure A). The application of the assured method in the Middelburg Study indicated how environmental geohydrological decisions can be made and a groundwater yield determined that can be managed in a sustainable way (Annexure A). In environmental decision-making, risk must not be avoided or minimised, but rather identified, characterised, quantified or qualified and then managed.

The value of the assured model for decision-making in complex environmental problems is demonstrated by an application on these two additional case studies.

7.3 The decision-making envelope

Based on the findings of this study, a decision-making envelope is produced that indicates the convergent nature of the decision-making process. At least two information points from field studies or analogue information are required to determine the initial envelope in the decision-making process (Figure 7-4). From these starting points, a linear interpolation can be made on a semi-log plot where time and cost are plotted on the x-axis and information on the y-axis (Section 3). If at least two data points or clusters are available, the minimum and maximum of each of these can be extrapolated with a straight line. The uncertainty is then indicated by the angle of the envelope. As the process grows and more information becomes available, it converges. The decision-maker must determine when information is sufficient, based on the requirements of the investigation and the consequence of the decision that has to be made. The envelope also shows that it would require an infinite number of data points to reach zero uncertainty, which is an ideal, but impossible case, represented by a straight line (Section 3), Figure 7-4).

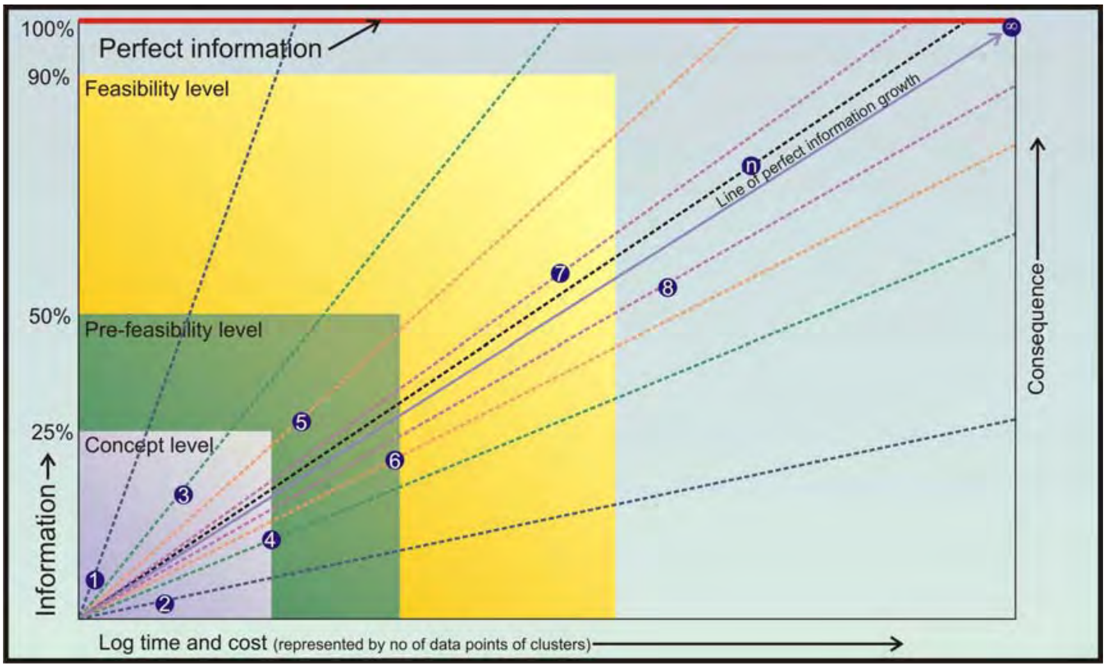


Figure 7-4 Schematic representation of the convergent decision-making process with decision-making envelopes.

The only discipline that could be found where a decision-making process is followed is in project management of large engineering projects, such as the development of new mines or infra-structure. The decision-making process in these studies is guided by project management principles where the project is defined from concept, pre-feasibility, feasibility and implementation phases. These phases are based on levels of uncertainty that are linked to costs. It will typically start with a concept study that has an accuracy of ±25%[63] in terms of cost. If the project is considered feasible, the level of accuracy is up-scaled to pre-feasibility level at an accuracy of say 50% and then to 90% for feasibility level after which implementation will take place. It is accepted that it is not practically possible to aim and obtain more than e.g. 90% accuracy and no effort is made to reach e.g. 100% assurance. If these processes can be followed in one of the spheres of sustainability such as the technology and development of large engineering projects, it should be expanded to include the other spheres of sustainability such as complex environmental problems to allow for sustainable development.

[63] The level of accuracy of each level is determined by the client or the project manager.

CHAPTER 8

"Not making a decision is also a decision" J. Maxwell.

8 CONCLUSIONS AND RECOMMENDATIONS

8.1 Introduction

This study was inspired by critical questions and input that surfaced during a number of investigations into complex environmental geohydrological problems. The critical questions and input arose mostly from scientists and were related to the adequacy of data. The requirements were for more and more data and as more data became available, more unknowns were uncovered, which increased uncertainties. The process followed a divergent trend that led to a lack in decision-making as it fostered doubt with decision-makers. This continued while business and political decision-makers would often ask the question: Now what is the status? Or what are the solutions? The divergent output was mostly based on a seemingly pure scientific approach that focused on obtaining data in the pursuit of information. The process can be equated to climbing a mountain. The higher one climbs, the further you see but less and less is recognisable on the horizon that shifts further and further away.

The aim of this study was to determine how the decision-making process functions based on the philosophy and principles that guides it. The study focuses on decision-making in relation to the collection of data and how it is translated into information for complex environmental management problems. Decision-making is evaluated to determine whether it is possible to obtain a holistic approach within the principles and constraints of sustainability. The principles of sustainability must account for its components, which are; technology and development, the environment and social (business, legal and political) aspects. The possibility of making decisions based on limited, sparse and even analogue data is evaluated. The role of data and information in the decision-making process is characterised with specific reference to the *sufficiency* of information, within time and financial constraints.

The decision-making process is characterised, which is used to formulate an assured decision-making methodology. This methodology makes use of an iterative systems process to be able to make sustainable decisions in complex environmental management problems, using sparse data that is associated with a high level of uncertainty.

The objectives of this study and how it was addressed in this thesis are described below:

1. What is decision-making and how does it relate to sustainability? A literature review was done in Chapter 2 on the meaning and theory of decision-making in the context of sustainability. In sustainability, the constraints of the environment, technology (development) and social components (economic, legal and political) were researched. Of particular importance was to determine what controls decision-making in each of these spheres and how it can be used in environmental management.
2. Is there a relationship between data and information in the decision-making process? In Chapter 3, the meaning of data and information was evaluated in the context of decision-making. Groundwater data from three field sites were used to analyse and characterise the relationship between data and information in the decision-making process.
3. Is more data better and when is information sufficient for decision-making? The sufficiency of data as it relates to information in the decision-making process was determined in Chapter 3. It was determined that information converges with the number of data points and that more data provides less and less information. A point of sufficiency is reached before an optimal point is reached. Beyond that point, more data does not provide meaningful information for decision-making although the cost of data is still the same.
4. If groundwater is used as an example of an environmental component, what are the characteristics of these parameters and how does it influence decision-making? Groundwater data and parameters were characterised in Chapter 4 in terms of uncertainty and how it can influence decision-making. The findings showed that groundwater data is sparse and associated with a high degree of uncertainty. It is difficult to use groundwater parameters in environmental decision-making if the correct framework is not used.
5. Can a systems approach be used for characterization, understanding and decision-making in complex environmental water management problems? Systems thinking and theory was described in Chapter 5. The applicability of systems modelling in decision-making on environmental water management was evaluated.

6. Are models accurate enough and does it add value to the decision-making process? The purpose and application of models within the context of decision-making was evaluated in Chapter 5.
7. Is it possible to apply a systems, modelling approach on a case study that is based on a complex environmental water management problem? In Chapter 6, the complex environmental water management problem of the mine flooding in the Far East Rand Basin (FERB) was analysed based on a systems modelling approach. The application of a systems modelling methodology proved to be important as it related to the components of sustainability in the process of reaching conclusions on management and mitigation measures. The systems approach was also applied on a water supply problem at Middelburg in the Eastern Cape in Annexure A to demonstrate its applicability in a different environmental water management setting.
8. Is it possible to develop an assured decision-making methodology that can be used even if data is sparse and associated with a high degree of uncertainty? In Chapter 7, an assured systems management model for environmental decision-making was developed based on the two case studies. The methodology can be used in decision-making in complex environmental problems where data is sparse and associated with a high degree of uncertainty.

8.2 Conclusions

Decision-making is one of the most important aspects of everyday life. The theory of decision-making was historically developed in warfare and business management. A decision entails making a choice, judgement or to come to a conclusion about a matter. Decision-making is an increasingly important requirement in complex environmental problems. Recent examples are the rising acid mine water below Johannesburg and the BP oil spill in the Gulf of Mexico where a lack of decision-making based on the principles of sustainability was responsible for regional and far reaching environmental problems. Decision-making is the process of choosing the best alternative from a number of given options in terms of gains and losses, or to develop new options that may not yet exist.

Decision-making can be classified into three basic levels of strategic, tactical and innovative decision-making. Strategic decision-making is based on a holistic approach that is applied on a high level and includes the planning phase where a general direction is chosen without the consideration of detail. Tactical decision-making is based on a reductionist approach that is concerned with an analysis of the detail and sequencing within the boundaries of the strategy. Innovative decision-making is required where no suitable options or alternatives exist. Most of the decision-making problems are encountered when a defined process is not followed. Intuitive decision-making based on past experience alone, which does not make use of strategic planning, tactical implementation and evaluation of information leads to decision failures.

Decision-making in environmental management problems must be based on the triple bottom line of sustainability. The triple bottom line takes a holistic approach and balances the environment, development and social aspects (or people, planet and profit). The diverse nature of the components of sustainability creates an invisible but opposite pressure that counteracts sustainability. Environmental problems are generated or exacerbated when decision-making is applied outside the principles of sustainability.

In business management, there are two decision-making methods, namely the rational and the bounded rationality methods. The rational method is a step wise formal set of actions that seeks to evaluate all possible options with advantages and disadvantages to find the *best* or *optimal* outcome. It was found that the rational method is generally not followed in business decision-making, instead, the bounded rationality method is more commonly used. Due to time and budget limitations, the bounded rational method aims at making *near-optimal based on sufficient information* rather than *optimal* decisions that require a lot of information.

Environmental regulation is done through legislative processes. It is therefore important to determine how and on what decision-making is based in judicial processes. There are two main types of legal processes, namely criminal and civil procedures. Criminal cases require *proof beyond reasonable doubt* while in civil cases, there is a *gradation of proof*. Environmental violators can be prosecuted on either or both of these procedures. Practical applications have shown that judicial proof is different than mathematical proof and that scientific investigations are challenged in this way by judicial processes. Although probability theory used in mathematics is useful, there are numerous cases where it failed to be used as judicial proof. Judicial decision-making is based on evidence, values and principles and provide important considerations for environmental decision-making. Environmental aspects and sustainable development started to receive political and social attention in the past decade. Political motives are well developed and legislated in South Africa. In most cases environmental decision-making and authorization is not motivated by technical, but by social and political aspects.

The origin and basis of science is in philosophy. The scientific method is based on deductive and inductive reasoning. The definition whether something is scientific was defined by Karl Popper as a theory is scientific if it is *falsifiable.* He also defined science based on this definition as a *normal science*. There is no *exact science*, a terminology that is often misunderstood by scientists. The science of environmental and water management is based on probability theory. The Bayesian probability approach makes use of prior or analogue data that is iteratively refined as new data becomes available. The Bayesian approach is suitable for analysis of problems with limited data and a high degree of uncertainty and is widely used in environmental and groundwater data analyses. Science has powerful analysis tools, such as statistical methods, the decision tree, artificial intelligence, game theory, scientific models, risk assessments and optimization. A problematic aspect of science is that it cannot be used for decision-making when contradictions occur. Contradictions occur frequently in environmental management where choices based on trade-offs have to be made.

Environmental management is not the management of the environment. It is the management of the impact of humans on the environment using legal and scientific processes. Science is reductionist and seeks to differentiate into the detail while management is holistic and seeks to integrate. Decision-making is an integration of science for measurement and management to get people to plan and act. Decision-making in environmental management is based on legal processes. Effective environmental decision-making should be based on a strategic and holistic approach that is differentiated into the tactical and detailed reductionist approach in a balanced process. This balanced process must make provision for legal, scientific, social and business constraints.

Decision-making depends on information, which in turn is formed by data. Data that is analysed becomes information which is interpreted to become knowledge that can be used for decision-making. There is an incorrect reductionist scientific view that more and more data on increasing smaller scales would provide more information. Information is defined as an accumulation, arrangement and meaning of data. Data is considered as analogue to pixels that build a picture, that represents information. An accumulation of pixels does not necessarily build a picture. It is the accumulation and arrangement of pixels (or data) that provides a picture (or information). Once the picture can be recognised, the pixels or data is considered as *sufficient*. Additional pixels may not add value in terms of decision-making, but may still be collected at a cost if the decision-making process is not understood. *Perfect data* as an idealised concept was defined as data with a standard deviation of zero. With perfect data, only one data point is necessary to be able to determine the value of that specific parameter (e.g. depth to groundwater) across the domain of interest. The level of information was defined practically as the difference that each new data point contributes to the quantification of a physical parameter.

The study indicated that information vs. data follows a logarithmic trend that adheres to the *law of diminishing returns*. The information trend converges towards an asymptotic maximum that it never reaches (i.e. a straight line on a semi-log graph) and it provides less and less information with an accumulation of data. The field data indicated that if the analyst would accept a specific error range, such as 5% as acceptable, then minimal data of only 5 to 10 data points could provide *sufficient* information. The data gathering process reaches an *optimal* point which concludes that *more data is not necessarily better*. It is therefore *not scientific* to intuitively expect that more and more data on smaller and smaller scales would be more *accurate*. The collection of data is associated with *cost* and *time*. Data sufficiency and financial constraints should determine when data collection should end. It was determined that there is also an *optimal* point in terms of cost and information obtained. Beyond that point no significant *data worth* is obtained to warrant additional data collection. To obtain *perfect information* is not possible as it would be at an infinite time and cost. Perfect information was defined by a straight horizontal line on the linear plot of information vs time and cost. A comparison of the information curves that were obtained from the three field sites indicated that the slope on a semi-log plot differed. More information is associated per additional data point along curves that have steeper slopes (Figure 8-1). The amount of time and cost that should be spent to follow the information curve must depend on the *risk* or *consequence* of the project for which the decision must be taken. It was shown that if more information is added than required, it could obstruct the decision-making process. This is known as the *less-is-more-effe*ct, which is an antidote to the *analysis-paralysis* problem of data gathering and interpretation that is often experienced in scientific studies of complex environmental problems.

Assumptions are used in the decision-making process to substitute information. Assumptions are usually criticised in scientific studies but in the absence of *perfect information*, which is always, assumptions have to be made. As more information becomes available, the analyst has to rely on assumptions on a lesser degree, but it can never be ruled out completely. It was found that it is not the assumption that is the problem, but in the way it is used. Assumptions in environmental decision-making must be made in line with the *precautionary principle* used in judicial processes and the *minimax* approach used in game theory. The minimax philosophy is a strategy that originated from game theory and is chosen to limit or even remove uncertainty from the decision-making process.

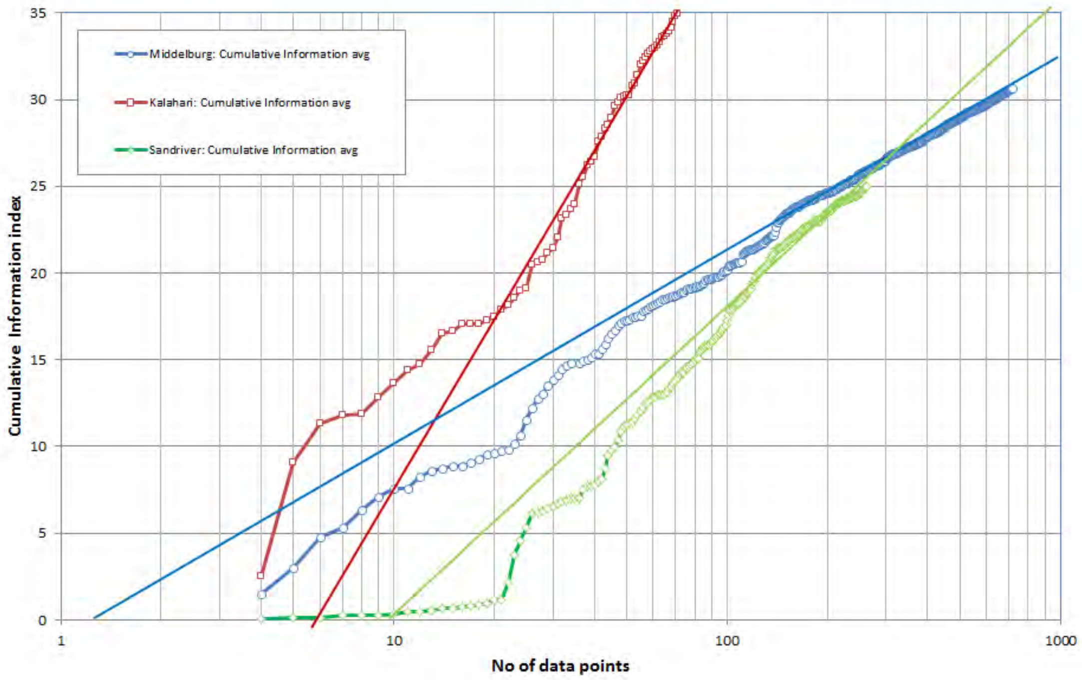


Figure 8-1 Middelburg, Kalahari and Sand River sites information curves compared.

The characteristics of groundwater data were investigated as an environmental component to determine which of the data and parameters can be considered as so called *exact* data. It was found that groundwater data is spatially variable, sparse and associated with a high degree of uncertainty. The assessment indicated that of the 9 main groundwater variables, only 3 (water levels, abstraction rates and water quality) can be measured. Most of the other parameters are indirectly derived from field tests based on analytical models that make use of idealised assumptions or based on qualified estimates. Geohydrology is therefore *not an exact science* with *no unique* data sets or solutions. These aspects need to be taken into account during site characterization programmes for the purposes of decision-making and should apply specifically to the Water Use Licensing and Reserve Determination Process.

Intuitive decision-making has proven unsuccessful, especially when non-linear problems are dealt with. Systems thinking have proven beneficial in the understanding of problems where complex systems influence each other as a whole. The systems view differs from the historical viewpoint of linear systems with cause and effect in that it consists of complex systems with feedback. A systems approach is ideal for evaluation of complex environmental problems e.g. the hydrological cycle in nature behaves like a system. Models, and more specifically systems models, are increasingly used as scientific tools that provide valuable, unseen information in the decision process. Models turn data into information and are used to assist the analyst to *see beyond the horizon*. Models in the context of decision-making should primarily not be used to aim and simulate the physical system as accurately as possible (this is the aim of models in research). It should be used to *model the problem*. The *purpose* of the model is important as models that are used for the wrong purpose or context create problems in terms of expectations as to what can be deduced based on the model output. A specific problem that was identified is the expectation by analysts who believe from what they see or experience that some scientific things are just plain true or false. These are known as *naive realists* who believe that this should apply to the use of models. Problematic misuse of models is to use it to try and *predict the future*. Models should rather be used in *prospective* evaluations of predefined *scenarios*, which aims to determine the influence of variables on future performance and does not try to predict the future. *Scenario development* and testing is one of the important steps of the modelling and decision-making process. A framework was developed within which systems models can be used for the purposes of environmental decision-making.

The systems model approach was applied on a complex current environmental water management problem of the Far East Rand Basin (FERB) flooding and decanting. It was used within a systems based holistic decision-making framework that demonstrated how a complex environmental problem can be evaluated and decisions reached on management and mitigation measures.

From this study, an *assured model for environmental decision-making* was developed. The model follows an iterative process that is based on the bounded rationality model for decision-making. It requires a definition of the problem that bounds it and that determines the purpose of the assessment. It is a decision-making methodology based on the principles of sustainability, which makes provision for business, development (technology) and social (legal and political) aspects and constraints. The process is based on strategy mapping, data analysis, model development, scenario development and testing, a critical evaluation of results and converges towards decision-making. Important and strategic principles in the process are that a *case is built for and against each scenario* and when assumptions are made, the judicial *precautionary principle* and the principles of *falsifiability* and *minimax* are applied as decision rules. This is due to the fact that the decision-maker accepts that the actual answer is and will be unknown or even unknowable, but that the decision is made based on sufficient information that would still result in having a positive outcome, even if assumptions are wrong.

Criticism to the assured method for environmental decision-making could be that due to the application of the precautionary and falsifiability principles, it could create practical problems by being perceived as *risk-averse*. Due to the *minimax* approach followed in the *assured method*, it could be perceived as not providing *the correct answer*. The application of the assured method seeks to determine a *minimum or maximum* value that can be used for decision-making, as it is accepted that the actual value is unknown. By aiming to determine the actual value in most environmental problems, it makes it impossible for regulators to reach or make decisions as any output could easily be contested in court and debated for years as the *actual* answer varies with time and it is most cases unknowable. What can be determined is a *minimum* or a *maximum* which can be tracked with time to converge in a *decision-making envelope*. Based on the convergent nature of this process the decision-maker must determine when the uncertainty in the decision-making envelope is acceptable. This is usually based on the risk or consequence of the project. Field applications of the assured method have shown that it can be used to make non-conservative decisions where risk is controlled by a systems approach with decision-making rules and time frames. Decision-making involves taking of qualified risks. In environmental decision-making, risk must not be avoided or minimised, but rather identified, characterised, quantified or qualified and then *managed*.

From this study, it is the experience that when it comes to decision-making, most scientists are unprepared and tend to be *risk-averse*. Decisions that have to be made where data is limited and uncertainty high are usually done based on a risk-averse or risk-avoidance approach. This violates the principle of sustainability as it scales the cost on the highest possible end, which has a significant effect, because the relationship between risk and cost is exponential. The scientific approach sowed doubt for the political decision-makers in the Far East Rand Basin flooding problem, which exacerbated indecision by government. Was it then not an incorrect approach to ask scientists to make decisions on an environmental management problem? This has in part led to the discouragement of political will to act, as the *seemingly pure scientific* approach provided more questions than answers during the project, which led to a divergent decision-making process. The Water Use Licensing process is another example of a scientific data gathering process that should have been a decision-making process. The result is indecision by the regulatory authority.

This study showed that there is no *scientific* basis to strive for obtaining exact information by obtaining more data based on more and more scientific studies. It is not scientific to expect *perfect information* before decisions can be made. Are we as scientists taught to perceive science as exact and data as the potential answer to the lack of information? Are scientists taught that the mechanics of science are exact and that everything should be experimentally proven? The basis of science is philosophy that does not rely on an exact physical world, but rather accepts the usefulness of the philosophical, variable, statistical and hypothetical deductions that can be made. It must be understood by all students of science that it is an exploratory field in which the unknown is pursued on the basis of hypotheses and assumptions. Decision-making is therefore a management process that makes use of scientific tools for the purposes of quantification and provision of data. Input for decision-makers should not be done by scientists alone, but by specialist teams that provide the data, which must be interpreted by decision-making experts that have the capability to interpret the information from the data and reach decisions by following appropriate processes such as the assured method for environmental decision-making.

Where science is in the pursuit of *knowledge*, environmental decision-making should be based on *wisdom*[64]. In the past, numerous environmental disasters were based on the application of science and sound knowledge without wisdom. The recent oil spill of the BP Macondo Well on the Deepwater Horizon Drilling Rig is an example of how knowledge of deep seated oil and gas in the Gulf of Mexico was used to explore it without the wisdom of pre-empting the downstream safety and environmental impacts and controls that were required for such an exercise (National Geographic, 2010).

The decision-making process that was characterised in this study would have applications beyond the environmental sphere and could be applicable in other disciplines.

8.3 Recommendations

Based on the findings of this study, the following recommendations are made:

- Research should be done on the effect that scientific approaches have on political decision-making.
- Scientists should be trained in basic decision-making principles and processes.
- Scientists should be trained to understand the philosophy and origin of science and the fact that there is no definition or existence of an *exact science*.
- Additional applications and testing of the assured model for environmental decision-making should be done on field cases.
- The assured method for decision-making should be evaluated for application in other disciplines, such as environmental regulation, engineering, business development, conservation etc.
- The definition of sustainable yield in groundwater should be revised to apply to the principles of sustainability.
- The water use licensing process should be revised based on decision-making principles rather than the current data gathering process that leads to indecision.
- Additional research should be done to determine the characteristics and what influences

[64] The ability to discern or judge what is true, right, or lasting; insight. Common sense; good judgment. The sum of learning through the ages; knowledge (http://www.thefreedictionary.com/wisdom). Wisdom is when knowledge is applied within the correct context.

the information gained and the slope of the information curves that were obtained in the field studies (Figure 8-1).

"The wise also will hear and increase in learning, and the person of understanding will acquire skill and attain to sound counsel so that he may be able to steer his course rightly "Proverbs. 1:5. Amplified Bible.

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ANNEXURE A

10 ANNEXURE A: APPLICATION OF THE ASSURED SYSTEMS METHOD ON THE MIDDELBURG GROUNDWATER SUPPLY PROBLEM

10.1 Introduction

The supply of groundwater to Middelburg Town in the Eastern Cape Province of South Africa (Figure 3-3) must be done based on an approach that would ensure sustainability of supply. This poses a challenge in the face of the fact that the existing Municipal Aquifer has proven to be unsustainable. The immediate water demand for Middelburg is 3500 m^3/d (40 ℓ/s) with the purpose of the investigation (AGES, 2010c) to supply in the immediate demand and to provide options for future demand.

Apart from the existing Municipal Aquifer, there are no long-term water level data available for the regional groundwater system. This poses a problem for regulatory decision-makers as the groundwater supply is associated with uncertainties regarding recharge from rainfall, transmissivity, storativity, etc. (Section 4). The quest for sustainable groundwater supply equates to steady-state supply where time does not play a role, which may not seem practicable or even possible, given the uncertainties. This is due to the large number of unknowns and variables that influences the decision-making process. These variables include land use changes, geological uncertainty and even changes in rainfall patterns that influence the sustainability of groundwater. Some of these parameters like recharge and storativity are qualified guesses at best (Section 4).

The question is how do we develop and manage such a resource? There are two options; the first is to accept that groundwater is associated with uncertainties and to follow a development and management approach that caters for the groundwater characteristics. The second is to conclude that groundwater is such an uncertain resource that we should abandon it and look for other alternatives. The second option is, according to this research, not an option as will be discussed in this section. It is accepted that groundwater resource quantification techniques such as modelling (and statistics for that matter) are all wrong, but still the most useful options that exist (Poeter, 2006).

In this section the approach to groundwater resource development and management based on the assured systems method is applied.

10.2 Application of the assured systems method to groundwater development

The groundwater development plan was done based on a phased approach as follows:

1. The existing groundwater data (borehole distribution, water levels, rainfall, water quality) was evaluated in terms of groundwater performance and history. It is known that the Municipal Aquifer is over utilised as water levels declined in the past 20 years to levels that could not sustain the required abstraction rate of 83 ℓ/s to 100 ℓ/s. The decline in water levels equates to abstraction that exceeded the natural recharge and the surplus volume had to be obtained from storage until it was partially depleted. It was also found that the abstraction at the Municipal Aquifer is focused in one specific area (AGES, 2010c).

2. A regional hydrocensus was done to determine the groundwater status in terms of distribution of the regional boreholes, water levels and abstraction rates.

3. Based on the minimax approach of the *assured systems method for decision-making* (Section 7.2), a regional *minimum groundwater balance* was determined that was based on quaternary catchments and defined sub-catchments which were delineated based on the hydrogeological nature of identified groundwater development target areas. The purpose was to evaluate the potential for groundwater supply on a regional scale. It is noted that the groundwater balance approach, if applied incorrectly, can lead to unsustainable groundwater abstraction evaluations (Bredehoeft et al. 1982; Devlin and Sophocleous, 2005, Seward et al. 2006 and Zhou, 2009).

    a. The *minimum groundwater balance* approach was applied here, which is different from the classic groundwater balance approach in that the aim was not to determine the *actual or natural groundwater balance* based on natural recharge, but a *minimum groundwater balance* based on assured rainfall, for the purposes of planning for water supply (Vivier et al. 2009b). It is accepted that the actual groundwater balance is unknown and that there are too many uncertain parameters at this stage of the investigation to aim and identify it.

    b. In the absence of actual groundwater information (which costs money to obtain), conservative assumptions were made (Section 3.7). The assumptions were conservative in that the MAP was not considered for recharge estimations, but the lower 95th percentile (which represents a

1:20 year drought) and aquifer storativity was neglected at this stage.

c. The groundwater balance values were then determined for smaller surface water catchments to identify the groundwater supply potential. These smaller sub-catchments were delineated based on geological structures such as dolerite rings as well as dolerite dyke intrusions that were targeted for well-field development. The reason for this approach is that the actual groundwater catchment is unknown. The assured recharge, existing abstraction, discharge to springs and losses to evaporation was considered as the *minimum* in the groundwater balance approach. This means that there could be more but not less groundwater available than the outcome of this investigation shows. The groundwater balance indicated that new groundwater resource development could be possible in three new undeveloped aquifer areas such as Dunblane, The Glen, Karmel and Lusernvlei Aquifer Areas. High volume production boreholes are absent in these identified target areas and the assessment was done based only on surface water catchments and recharge potential based on the lower 95^{th} percentile. It must be noted that the MAP is 350 mm/a and the lower 95^{th} percentile is 180 mm/a, which is only 52% of the MAP (i.e. conservative). Analogue information on regional geological structures such as regional linear dykes and ring dykes, further assisted in the predetermination of groundwater supply potential in terms of borehole yields.

d. The *assured approach* was used as a decision-making and planning tool to determine where it would be cost-efficient to spend money on drilling of new boreholes. It would be illogical to drill boreholes in areas where recharge is low or limited, abstraction or borehole density is already high or catchment areas are small. The minimum groundwater balance approach also considers economical considerations as sustainability is defined by the triple bottom line that includes development, environment and economy (Brundtland, 1987; Vivier, 2006).

e. The approach is based on a Bayesian methodology where prior information is used in the absence of actual data. The uncertainty with prior information is high and hence conservative assumptions are made. The information is re-assessed iteratively as field data becomes available until it converges to the analyst's satisfaction (Freeze et al. 1990). It is accepted that perfect data does not exist and that we will have to deal with uncertain information until we can reduce uncertainty with actual monitoring information (Vivier, et al. 2009c). Prior information includes only minimum groundwater balance values and regional geological information. Posterior information includes drilling and aquifer testing information.

4. The assured approach was used to focus further work. The areas that indicated potential for additional groundwater supply based on the groundwater balance evaluations were evaluated for local hydrogeological structures such as dolerite dykes and ring intrusions and associated fractured zones.

5. This was followed by the site investigation, exploration borehole drilling and aquifer testing which comprise the main portion of the project expenditure.

6. The aquifer tests were used to determine how much water can be abstracted successfully from boreholes. The principle of constraints was used (Goldratt, 1990) to determine the initial recommended abstraction rates. If the constraint is the borehole yield, then it was used as the limiting factor for water supply (and not the minimum groundwater balance figure) and if the constraint was the minimum recharge based on the 1:20 year drought, then it was used and the borehole yields were scaled down. Safety factors were included following the analysis of the aquifer tests to estimate the initial sustainable yields. At this stage, it is accepted that the actual groundwater recharge and the spatial extent of the borehole capture zone is unknown and that a more sustainable yield can only be determined through monitoring.

Following this methodology, a systems approach to sustainable groundwater development and management was followed for Middelburg.

10.3 Systems approach to groundwater development and management

The effect of uncertainties in groundwater development and management on the determination of sustainable yields is that it drives the recommended yields lower, in an attempt to minimise or avoid risk. This is the so called *risk-averse* approach that was used by review scientists in this project (Freeze et al. 1990). In this approach, the groundwater yields from boreholes are recommended at a very low rate to aim and obtain supply that can last virtually "forever" (steady-state approach). To obtain the required future supply of 40 ℓ/s and using the risk-averse approach, boreholes would be recommended at very low rates of typically 2 ℓ/s/borehole (Van Tonder, 2010) even if the pump tests indicated yields of up to 20 ℓ/s or 10% of the tested yields (Figure 10-1).

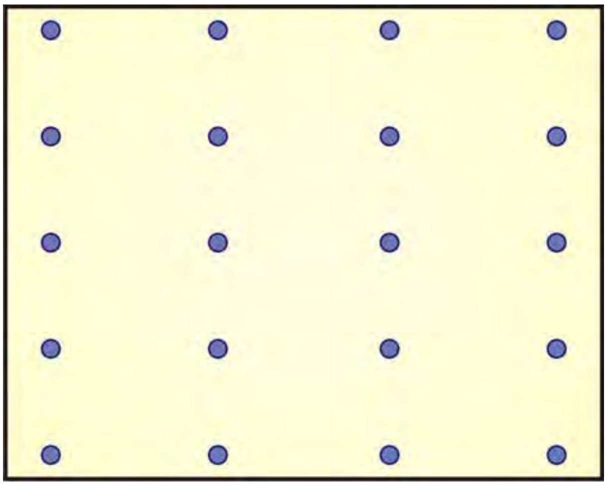

Figure 10-1 Sustainability from a steady-state approach (40 ℓ/s at 2 ℓ/s/borehole = 20 boreholes).

The problem with this approach is that due to the risk-cost relationship, it drives costs up (Figure 2-21). The engineering design of infra-structure has to take cognisance of not only risk but also cost. Due to the fact that sustainability is not only defined by environmental considerations, but also by development and the economy (Brundtland, 1987), sustainable yield is not the lowest possible yield.

The terminology used for sustainable yield in the groundwater papers evaluated in this investigation (Seward et al. 2006 and Zhou, 2009) neglects development and economical effects and only concentrates on environmental geohydrological requirements. Following these approaches would lead to the most expensive groundwater development possible (Figure 2-1). Sustainable yield in terms of the definition of sustainability (Section 2.5) can only be defined if economic considerations were accounted for. Another terminology (such as environmental safe yield) will have to be used for what is meant with sustainable yield in the papers.

For reference purposes, at Middelburg, the surface water option is estimated to be in the order of R300 million capital cost and a high operating energy cost as water has to be lifted at 300 m head and pumped across a long distance.

Annexure A: Application of the assured model on the Middelburg Site

A conceptual calculation was done for groundwater development at Middelburg using estimated groundwater development and infra-structure capital costing with the same reference (Table 10-1). Operational costs were neglected to simplify the comparison. The conceptual costing assumed that there are a number of options for groundwater development that ranges from extreme ends by using 2 boreholes at 20 ℓ/s/borehole which is associated with a high risk (i.e. risky) to using 20 boreholes at 2 ℓ/s/borehole associated with very low risk (risk-averse) of failure (Table 10-1). The unit costs were assumed to be the same. The cost graph indicates an exponential increase in costs with lowering in abstraction rate per borehole (Figure 10-2).

Table 10-1 Middelburg: Conceptual comparative costs for groundwater development following various options.

No	No of boreholes	Yield per borehole (ℓ/s/borehole)	Cost of boreholes (R mil)	Cost of infra-structure (R mil)	Total capital cost (R mil)	Total yield (ℓ/s)	Unit capital cost over 20 year period (R/m^3)
1	2	20	R 0.60	R 10.00	R 10.60	40	R 0.42
2	4	10	R 1.20	R 20.00	R 21.20	40	R 0.84
3	8	5	R 2.40	R 40.00	R 42.40	40	R 1.68
4	16	2.5	R 4.80	R 80.00	R 84.80	40	R 3.36
5	20	2	R 6.00	R 100.00	R 106.00	40	R 4.20

The problem with the risky approach is that if the analyst is wrong, then the groundwater resource could be depleted but at a low infra-structure cost at R10.6 million. The problem with the risk-averse approach is that if the analyst is wrong and higher abstraction rates were possible then a total of R106 million of tax payers money was unnecessarily spent. In this, the risk-averse approach is more problematic than the risky approach in that if it is wrong, then the money cannot be retained, but in the risky approach, there are still options to expand the wellfields. If the middle road is taken, then groundwater can be supplied at around R 45 million.

The interest of the alternative of R300 million would amount to around R 30 million per year. The payback on the groundwater option would be 1.5 or in the worst case 3 years, which is a good financial option.

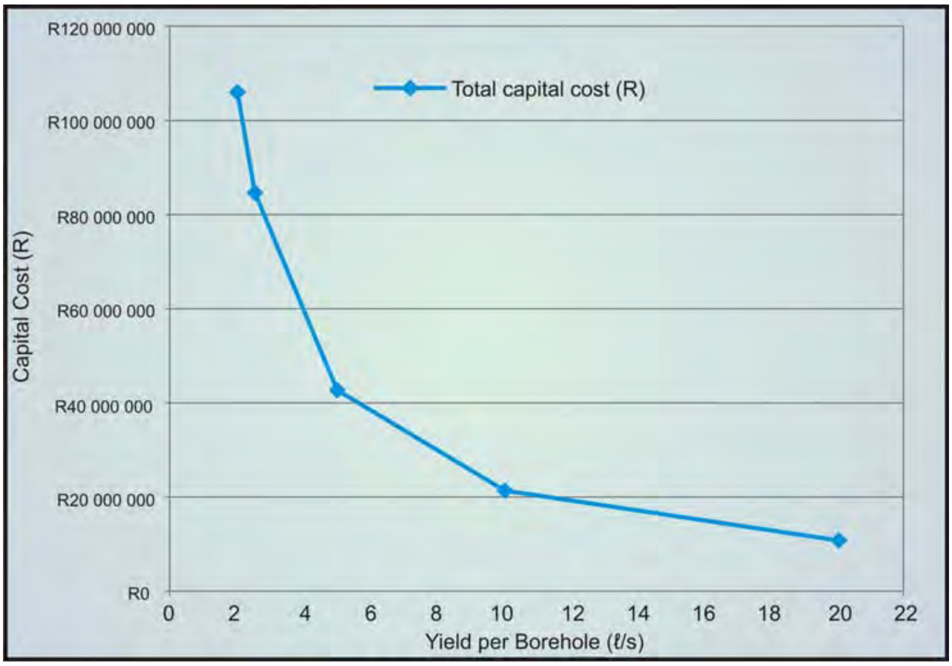

Figure 10-2 Middelburg groundwater development options vs cost.

By considering a systems approach to sustainable groundwater development and management, this problem can be overcome (Section 7.2.1). In systems theory, it is known that a problem can be exacerbated if the interactions between sub-systems are not understood (Sterman, 2002). If water is used liberally and leaks in the system are the main problem, then provision of more water would lead to more losses in the system. This is actually what happens at Middelburg, which from a systems viewpoint should be mitigated before any more water should be supplied.

The discussion will be confined to the groundwater supply problem to illustrate the application of the assured method to determine sustainable yield. With a systems approach, the water supply problem is considered as different but inter-dependant components and does not only consider groundwater and the environment but also economics. The problem is differentiated in terms of spatial and temporal components and does not aim to determine a sustainable yield per borehole that can last *forever* (Figure 10-3).

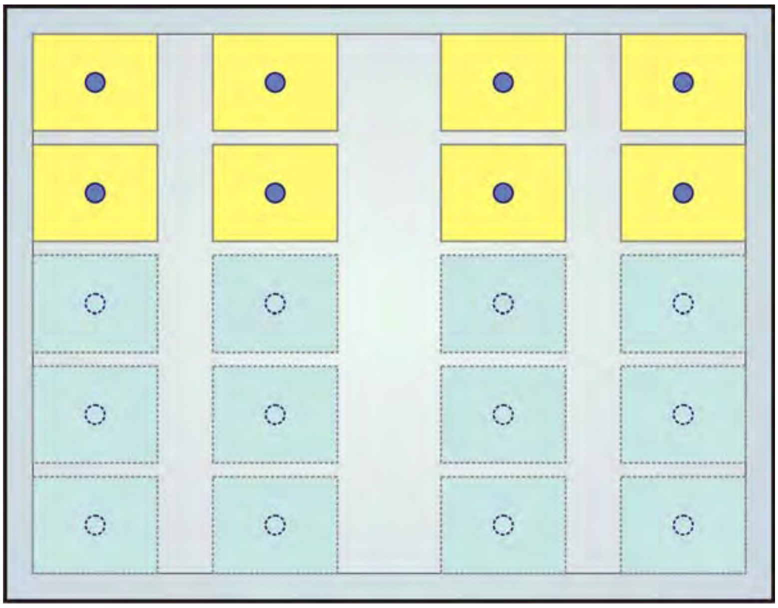

Figure 10-3 Systems approach to sustainable groundwater development and management at Middelburg (40 ℓ/s at 5 ℓ/s/borehole = 8 boreholes).

With this approach, the aquifer is developed at an initial solution that is expected to be near-optimal, but scaled on the higher risk and lower cost side of the curve (Figure 10-4). It is accepted that recharge, aquifer boundaries, geological interactions and storativity is unknown and that the minimum overall water is abstracted based on the minimum groundwater balance approach. Conservative qualified calculations and estimations were used to determine the initial conditions for abstraction. This would entail using a slightly higher initial abstraction rate for the boreholes that were tested. This abstraction rate of 8 ℓ/s is still significantly lower than the tested yield which was at 20 ℓ/s to 30 ℓ/s for actual field borehole tests (AGES, 2010c).

The approach entails that according to the level of understanding or knowledge, there are three different aquifer types at Middelburg:

1. The existing Municipal Aquifer with long-term monitoring of water levels and abstraction rates. High level of information/knowledge and high confidence. It is known that over-abstraction takes place.

2. New aquifers that were evaluated in terms of potential based on the *minimum groundwater balance* approach and where drilling and testing were done. The actual recharge, storativity and long-term groundwater behaviour is unknown and can only be determined through future monitoring.

3. New aquifers that were evaluated in terms of potential based on the minimum groundwater balance approach and where no drilling and testing were done. The groundwater potential indicates that it could be viable future options for groundwater development. The papers evaluated in this investigation that condemn the groundwater balance approach deny the opportunity to follow this step. This is due to the fact that these papers were compiled from an academic and research perspective that counteracts decision-making due to practical and economic considerations that were not considered (Bredehoeft et al. 1982; Seward et al. 2006).

The systems approach would be as follows (Figure 10-5; Table 10-2):

1. The aquifer would initially be operated at a near-optimal solution that is on the medium to high risk and medium to low cost side of the curve – for a limited period of time (Figure 10-4).

    a. The time limit for the recommended abstraction rates would be 1 to 2 years, which is a short enough period to prevent large scale over-abstraction and negative environmental impacts. In the case of Middelburg, this decision point would be to abstract 5 boreholes at 8 ℓ/s each to obtain 40 ℓ/s.

    b. Monitoring is done (using automated and manual systems) to prove or disprove the sustainability of the abstraction rates during this time.

2. If the abstraction rates are sustainable, then no unnecessary costs were incurred and the system can continue to operate on recommended abstraction rates. If not, the next one or two aquifer areas can be developed well in advance (1 to 2 years). The monitoring will be used as early warning systems to pre-detect excessive decline in groundwater heads with time before it can occur.

3. The abstraction rates at the initial two aquifers can then be reduced that would allow recovery once the third and fourth aquifers are developed and ready for use.

4. The same process can be re-iterated until the optimal (i.e. sustainable) solution is found with time.

By following this approach, the sustainable groundwater yields can be obtained iteratively with time without incurring undue high costs from the start. The decision principle that is used in the *assured method* is *what will happen if our assumptions are wrong* (Section 7.2)? In the assured systems approach, there are no negative consequences while the risk-averse approach could lead to an enormous waste of capital.

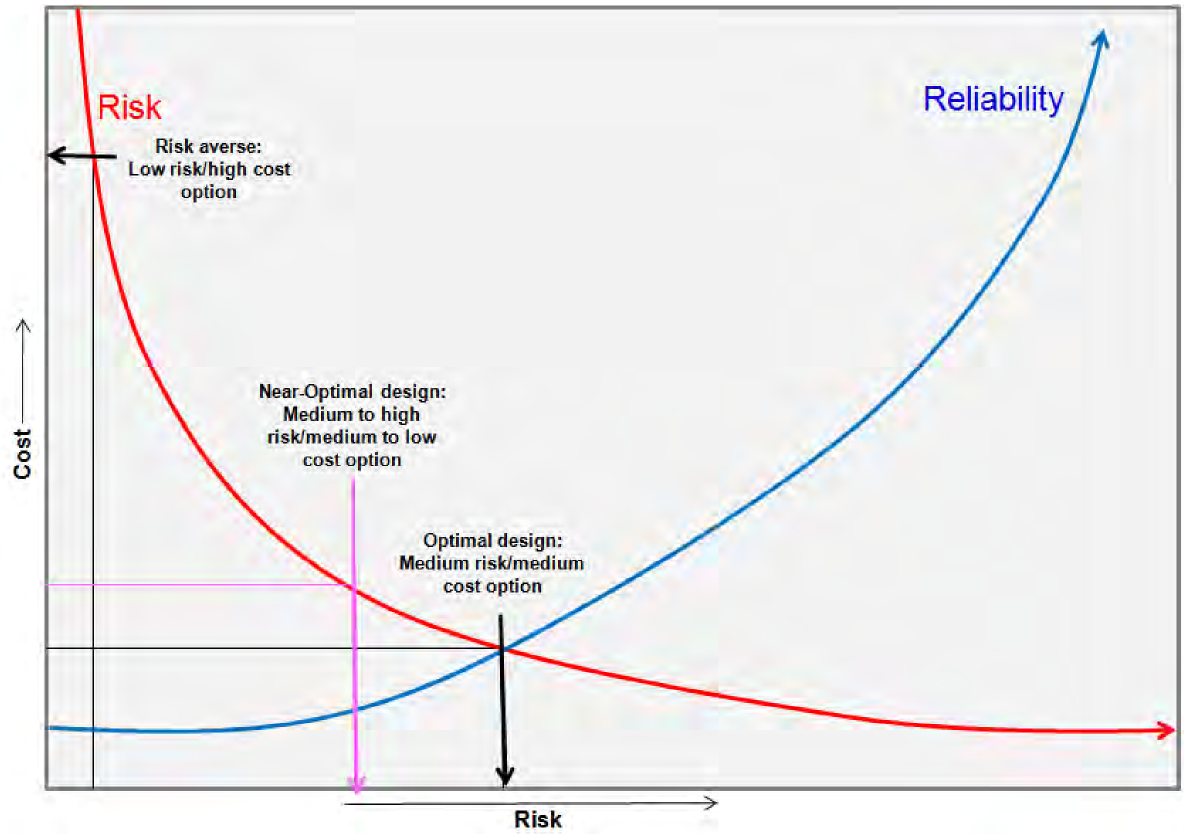

Figure 10-4 Risk-cost and reliability relationship with expected near optimal solution.

10.3.1 Middelburg groundwater development and management plan

The sustainability of groundwater at Middelburg will depend on how the resources are developed and managed. A phased and assured groundwater development and management plan was compiled based on a systems approach for sustainable management (Section 10.3). The purpose of the management plan is not to manage a single aquifer so that a decline in water levels would never occur, but rather the collective management of all the aquifers for a cumulative sustainable supply of the *system*. The sustainable management plan also accounts for the economics of the resource development and aim to prevent over-expenditure due to a risk-averse approach (Table 10-1, Figure 10-5; Table 10-2).

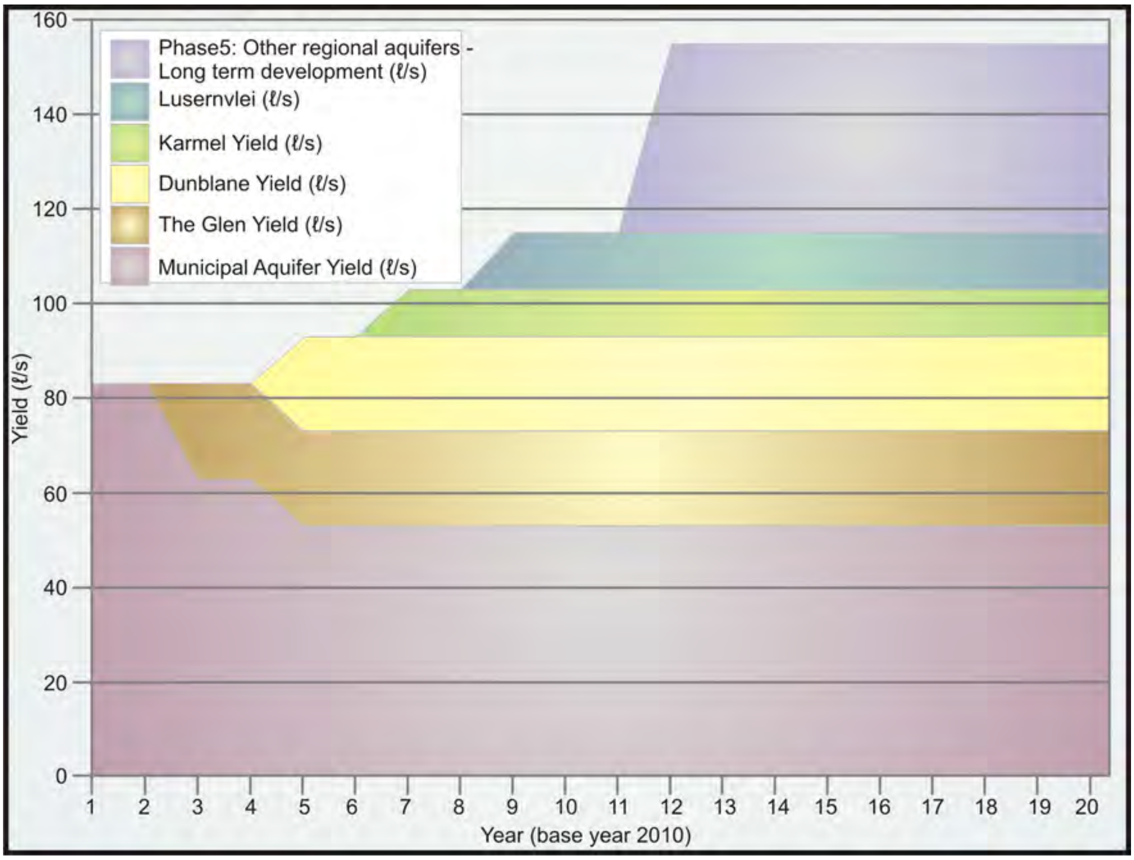

Figure 10-5 Middelburg groundwater development and management phases with yield.

Annexure A: Application of the assured model on the Middelburg Site

No	Description	Year 1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
No	Cumulative Yield (L/s)	83	83	83	83	93	93	103	103	115	115	115	155	155	155	155	155	155	155	155	155
0	**Phase 0: Base case: Existing Municipal Aquifer abstracting at 83 L/s**																				
0.1	Continue abstraction at existing Municipal Aquifer at 83 L/s for a period of 1-2 years until the new aquifer systems are activated																				
0.2	Reduce the abstraction at the existing Municipal Aquifer from 83 L/s to 63 L/s (25% reduction from current status). This should also allow the aquifer to recover to within the next 3-8 years																				
0.3	Reduce the abstraction at the existing Municipal Aquifer from 63 L/s to 53 L/s (35% reduction from current status). This should also allow the aquifer to recover to within the next 3-8 years																				
	Municipal Aquifer Yield (L/s)	83	83	63	63	53	53	53	53	53	53	53	53	53	53	53	53	53	53	53	53
1	**Phase 1: Development of The Glen Aquifer (103 L/s total supply)**																				
1.1	Abstract for 1-2 years while monitoring																				
1.2	Adust after 1 or 2 years if required																				
1.3	Continue monitoring and re-adjust if required																				
	The Glen Yield (L/s)			20	20	20	20	20	20	20	20	20	20	20	20	20	20	20	20	20	20
2	**Phase 2: Development of Dunblane Aquifer and reduction of Municipal Aquifer (103 L/s)**																				
2.1	Develop Dunblane Aquifer for 15 L/s and monitor for a period of 1-2 years																				
2.3	Continue monitoring and re-adjust if required																				
	Dunblane Yield (L/s)					20	20	20	20	20	20	20	20	20	20	20	20	20	20	20	20
3	**Phase 3: Development of Karmel Aquifer and reduction of Municipal Aquifer (103 L/s)**																				
3.1	Develop Karmel Aquifer for 10 L/s and monitoring for a period of 1-2 years																				
	Karmel Yield (L/s)							10	10	10	10	10	10	10	10	10	10	10	10	10	10
4	**Phase 4: Development of Lusernvlei Aquifer (123 L/s)**																				
4.1	Develop Lusernvlei Aquifer for 12 L/s and monitor for 1-2 years																				
4.2	Continue monitoring an re-adjust of required																				
	Lusernvlei (L/s)									12	12	12	12	12	12	12	12	12	12	12	12
5	**Phase 5: Other regional aquifers - long term development (40 L/s)**												40	40	40	40	40	40	40	40	40

Table 10-2 Middelburg sustainable groundwater development and management plan with phases.

10.4 References

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