

Evaluating the Vulnerability and Contamination of Groundwater from Mining Dump in Welkom, Free State, South Africa

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Declaration of Independent Work

I, **Silent Ruzvidzo**, identity number ##### and student number #####, hereby declare that this research project submitted to the Central University of Technology, Free State, for the degree of Doctor of Philosophy in Environmental Health, is my own independent work; and complies with the Code of Academic Integrity, as well as other relevant policies, procedures, rules and regulations of the Central University of Technology, Free State; and has not been submitted before to any institution by myself or any other person in fulfilment (or partial fulfilment) of the requirements for the attainment of any qualification.

SIGNATURE OF STUDENT

04/05/2024
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Abstract

The quality of groundwater in and around Welkom, a city in the Free State Province of South Africa, has been steadily declining over the years. This deterioration has led public and private institutions, including schools, churches, hospitals, and universities, to cease using groundwater for drinking due to health concerns. Previous studies have suggested that contaminants from mine dumps in the Welkom area are the primary cause of groundwater pollution. This study aimed to assess the vulnerability of groundwater to contamination from a specific mine dump in Welkom. A key objective of the research was to develop a point source vulnerability assessment method tailored to identify groundwater vulnerability to contamination from mine dumps. The study introduced a new parametric method called DWAPH, which considers five parameters: Depth to groundwater, Water quality, Aquifer type, Precipitation, and Horizontal distance to the contaminant source. The DWAPH method uses an index with four vulnerability scales: low (5-10), moderate (11-15), high (16-20), and very high (20-25). The study data, informed by the parameters was mainly gathered from five boreholes at the Central University of Technology campus in Welkom and a nearby mine dump. The boreholes were used as proxies to investigate the effects a nearby mine dump on groundwater quality and vulnerability to contamination. Each of the five boreholes was characterised through electrical conductivity and dilution tests to detect the fracture points and type of aquifers. The characterisation also involved the assessment of the depth to groundwater, and the depth of each of the five boreholes using a groundwater level meter. Each of the five boreholes, together with a nearby effluent from a mine stream, were seasonally sampled in the summer and autumn seasons between 2021 and 2023 and the water was tested for physical, chemical, and microbiological elements such as Electrical Conductivity (EC), Total Dissolved Solids (TDS), Dissolved Oxygen (DO), (Potential of Hydrogen) pH, Sodium(Na), Calcium (Ca), Magnesium (Mg) , Chloride (Cl), Fluoride (F), Sulphate (SO₄) and Nitrate (NO₃). The Weighted Arithmetic Index method was used to investigate the Water Quality Index using the same set of physical and chemical parameters. Further geological and hydrological characterisation of the study site was conducted through a literature review of previous studies. Results from the characterisation process indicated that all five boreholes were part of an unconfined

aquifer with fracture points detected at depths of 10 m for Boreholes 1, 2 and 5; 20 m for Borehole 3; and 24 m for Borehole 4. The results indicated that for Borehole 3, four parameters, EC, TDS, Na, and Cl, exceeded the South African National Standards (SANS 241), while all the other boreholes, except Borehole 1, had parameters that were within the recommended SANS 241 standard limits. Samples from the mine effluent stream had abnormally high levels of EC, TDS, DO, Na, Ca, Mg, Cl, F, SO₄ across all the months of the summer season. All boreholes, except Borehole 1 and Borehole 4, had total coliforms and faecal coliform counts that exceeded the SANS 241 drinking water guideline limits, while for the autumn season, the total coliform counts and the *E. coli* counts in all boreholes, except Borehole 3, were higher than the recommended SANS 241 drinking water guidelines. The results from the Water Quality Index (WQI) calculated through the Weighted Arithmetic Index (WAI) method using the SANS 241 water quality guidelines indicated that three of the five boreholes (B1, B3 and B5) had poor water quality, while only two boreholes (B2 and B4) had good water quality. The overall mean WQI of 68.16 that was calculated from the physical and chemical determinants of all five boreholes indicated that the water quality was poor and posed great health risks when used for drinking purposes. The WQI for the mine dump effluent was 544.31, which was five times beyond the minimum guideline requirements, hence supporting the possibility that it was heavily contaminated and can possibly contaminate nearby groundwater sources. The Welkom area was classified under the Witwatersrand supergroup with shale and sandstone. Using the background knowledge and data from the characterisation process, the DWAPH vulnerability assessment method was applied to assess the extent of groundwater vulnerability of the boreholes to contaminants from mine dumps. Results from the DWAPH index indicated an overall vulnerability score of 13 on a scale ranging from 5 to 25, which indicated moderate vulnerability. The outcomes from the DWAPH vulnerability method were compared against outcomes from the AVI, GOD, and RTt methods. Despite all three methods utilising the same datasets, their outcomes were not the same due to a variety of factors, which were mainly based on their design. The GOD and the DWAPH methods identified the extent of vulnerability of the Welkom area as moderate, while the AVI and RTt methods identified the extent of groundwater vulnerability as very high and low, respectively. The DWAPH method was further validated with NO₃- and bacteriological counts of total coliforms and *E. coli*. The nitrate validation indicated a weak correlation between nitrate concentrations and the DWAPH index of $R^2 = 0.0894$. The total coliform counts showed a weak correlation with

the DWAPH index while the *E. coli* counts indicated an almost perfect relationship with the outcomes from the DWAPH index. The preliminary findings of this study indicate that the groundwater in Welkom exhibits a moderate vulnerability to contamination from nearby mine dumps, with water quality tests revealing varying degrees of contamination. Given these results, it is imperative to exercise extreme caution when using groundwater in the area. Implementing restoration and treatment techniques, such as pump and treat, air stripping, filtration with granulated activated carbon, and air sparging, is strongly recommended before any groundwater is utilized. Future research in the Welkom area should prioritize using alternative parameters to further refine, consolidate, or validate the DWAPH method. Expanding the scope of groundwater assessments will provide a more comprehensive understanding of the contamination risks and help improve the reliability of vulnerability assessments. Additionally, it is crucial for local authorities and mining companies to collaborate in developing sustainable solutions to address the ongoing groundwater contamination issues. By working together, they can mitigate the environmental impact of mining activities and protect the health and well-being of the community.

Keywords: Vulnerability, Groundwater, Contamination, Assessment, Correlation.

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List of Acronyms and Abbreviations

AHP	Analytic hierarchy process
AMD	Acid Mine Drainage
ANN	Artificial Neural Networks
AVI	Aquifer Vulnerability Index
BOD	Biological Oxygen Demand
CCME	Canadian Council of Ministers of the Environment
CRG	Central Rand Group
CUT	Central University of Technology, Free State
DM	District Municipality
DO	Dissolved Oxygen
DRASTIC	Depth to groundwater, Recharge, Aquifer type, Soil texture, Topography, Impact of vadose zone, and hydraulic Conductivity
DRASTIC-L	Depth to groundwater, Recharge, Aquifer type, Soil texture, Topography, Impact of vadose zone, and hydraulic Conductivity – Land use
DRSTIC-LE	Depth to water table, net Recharge, Soil media, Topography, Impact of the vadose zone, hydraulic Conductivity, Land use type, and groundwater Exploitation
DRIST	Recharge, Impact of the vadose zone, Soil media and Topography
DWAF	Department of Water Affairs and Forestry
DWAPH	Direction of groundwater flow, Water quality, Aquifer type, Precipitation, Horizontal distance to contaminant source
EC	Electrical conductivity
AND	Acid mine drainage
<i>E. coli</i>	<i>Escherichia coli</i>
EW	Entropy weight
EW–AHP	Entropy weight–Analytic hierarchy process
GIS	Geographical Information Systems
GOD	Groundwater occurrence, overlaying strata, depth to groundwater

GODL	Groundwater confinement, aquifer overlying strata, depth to water table, and longitudinal conductance
HCS	Hydrogeologic complex and setting
Ksat	Saturated hydraulic conductivity
NSF	National Sanitation Foundation
PGVI	Pit Groundwater Vulnerability Index
PGVPI	Pit Groundwater Vulnerability to Pollution Index
RF	Random Forest
RTt	Rainfall travel-time
SANS	South African National Standard
SAWS	South African Weather Service
SI	Standard International
SINTACS	Water table depth (S), Effective infiltration (I), Unsaturated zone (N), Soil media (T), Aquifer media (A), Hydraulic conductivity zone (C), Topographic slope (S)
SSV	Soil screening value
TDS	Total dissolved solids
TSS	Total suspended solids
WAI	Weighted Arithmetic Index
WHO	World Health Organization
WNGM	Water flow and Nitrate transport Global Model
WQI	Water Quality Index
WRG	West Rand Group

List of Chemical Elements

As	Arsenic
Ca	Calcium
CaCO ₃	Calcium hardness
Cd	Cadmium
Cl	Chloride
CN ⁻	Cyanide
Co	Cobalt
Cr	Chromium
Cu	Copper
F	Fluoride
Fe	Iron
FeS ₂	Pyrite
FeSO ₄	Ferrous sulphate
H ₂ SO ₄	Sulphuric acid
Hg	Mercury
K	Potassium
Mg	Magnesium
Mn	Manganese
Mo	Molybdenum
Na	Sodium
NH ₄ ⁺	Ammonium
Ni	Nickel
NH ₄ ⁺	Ammonium
NO ₂ ⁻	Nitrite
NO ₃ ⁻	Nitrate
Mg	Magnesium
O ₂	Oxygen
Pb	Lead

PO_4	Phosphate
S	Sulphur
Se	Selenium
SO_4	Sulphate
SO_4^{2-}	Sulphate ion
Ur	Uranium
Zn	Zinc
Ur	Uranium
θ	Porosity
ρ	Electrical resistivity

List of Measurement Symbols

°C	Degrees Celsius
cm	centimetre
cm/s	centimetre per second
km	kilometre
kPa	Kilopascal
ℓ	Litre
ℓ/s	Litre per second
m	metre
mg/ℓ	Milligram/litre
mℓ	Millilitre
mm	Millimetre
mS/m	Millisiemens per metre
ppm	Parts per million
μg/ℓ	Micrograms per litre
μS/cm	Microsiemens per centimetre

Chapter 1

Introduction

1.1 Background of study

Groundwater is an essential source of drinking water in many sub-Saharan African countries (Misi et al., 2018). Traditionally, in South Africa, groundwater has been mainly used for drinking in rural and remote areas where the availability of tap water is limited. The accessibility, reliability, and generally good quality of groundwater has made it popular in areas where surface water is scarce and, in some instances, available but not of good quality (Burke and Villholth, 2007). However, in recent years, the increasing lack of drinkable water in most parts of South Africa due to the persistent droughts and fluctuating climatic conditions, has resulted in many local authorities struggling to provide reliable clean water to their communities (Ruzvidzo, 2019). This has led to many residents and institutions turning to groundwater as an alternative source of water (Ochieng et al., 2010).

In areas such as Welkom where communities live near abandoned mines and mine dumps, there have been several reports of groundwater contamination. Ebenebe (2016) found the concentrations of calcium (Ca) and sulphur (S) in water bodies within the Brandfort, Theunissen and Welkom areas as high as 100 ppm. A study conducted by Belle et al. (2021) revealed that close to 63% of the groundwater sources in the Lejweleputswa District Municipality (DM), covering the greater Welkom area, had poor drinking water quality. Figure 1.1 shows the water quality indices (WQI) from eight different boreholes across the Lejweleputswa DM.

Although findings from previous studies in the Lejweleputswa DM commonly indicate that the groundwater is of poor drinking quality, the source of contamination of the groundwater remains in speculation, as either from agriculture or mining activities. Further studies are therefore needed to confirm the source of groundwater pollution as well as to establish the extent of groundwater vulnerability in certain areas within the Lejweleputswa DM. This study will focus on assessing the vulnerability of groundwater contamination due to gold mine dumps in the Welkom area.



Figure 1.1 Water quality indices from boreholes across the Matjhabeng Local Municipality, Lejweleputswa District Municipality, Free State Province (Belle et al., 2021)

1.1.1 Mine dumps and groundwater pollution

Mining has been the cornerstone of the South African economy for many years. For over a century it has contributed considerably to the welfare of the people of South Africa. As a result of mining and associated industries, many settlements developed around mining areas, and these settlements gradually established into bigger towns and cities. The City of Welkom, located in the Matjhabeng Local Municipality, under the Lejweleputswa DM, was founded in 1949 because of the gold mining activities in the area and was declared a city in 1968 (Durand, 2012).

Although the mining activities have significantly supported the growth of the South African economy for centuries, there has been huge trade-offs with respect to environmental degradation in mining areas and communities as the mining activities have been responsible for environmental pollution at scalable proportions (Kamunda et al., 2016). This is mainly because the mining process involves the use of heavy metals when extracting minerals such as gold from its ore (Abdalla et al., 2010). Some of the by-products released from the extraction of gold from its ore include pyrite or iron disulphide, and heavy metals such as mercury (Hg), cyanide (CN⁻), lead (Pb), arsenic (As), and cadmium (Cd) (Zhou et al., 2016). Trace elements such as copper (Cu), chromium (Cr), manganese (Mn), nickel (Ni), zinc (Zn), molybdenum (Mo), and selenium (Se) have also been found as by-products of the mining process (Kurwadkar, 2017). The

contamination of groundwater resources with heavy metals has negative health implications for people living near mining areas and mine dumps (Hansen, 2015; Nkuba et al., 2019).

Huge amounts of waste by-products are produced during the processing of gold ores (Kossoff et al., 2014; Lottermoser, 2010). In some gold mines, where poor machinery are used and extraction procedures are compromised, a large percentage of the mined gold ends up as waste materials (Lottermoser, 2010; Ngure et al., 2014). These mining wastes are usually collected and accumulated in mine dumps, also referred to as tailings, on the surface of the earth surrounding the gold mines. The mine dumps comprise several contaminants, some of which come from the gold-containing ore, while others emerge from the processes of chemical extraction of the gold from its ore (Lottermoser, 2010). The contaminants contained in the mine dump can be easily dispersed through weather elements such as wind and rain. The surface run-off from the rain is the major driving force for groundwater contamination as it filtrates into aquifers below the earth's surface.

As the rainwater flows through mine dumps and infiltrates the ground, the sulphur (S) bearing pyrite reacts with air and water to form sulphuric acid (H_2SO_4) and ferrous sulphate ($FeSO_4$) (Kinna, 2016; McCarthy, 2011). The resulting acid surface run-off can further dissolve other trace elements before infiltrating the ground, thereby hugely contaminating the groundwater. The formation of H_2SO_4 from surface run-off and its subsequent contamination of surface water and groundwater is generally known as acid mine drainage (AMD) (Belle et al., 2021). AMD has disastrous consequences and poses severe health risks to the environment (Brahmi et al., 2021).

A study conducted by Naicker et al. (2003) discovered that the groundwater in some Johannesburg areas was acidified and severely contaminated due to the oxidation of pyrite confined in the mine dumps. The mine dumps also had significantly high heavy metal concentrations. Wade et al. (2002) noticed that when the groundwater table is close to the surface, the soil profiles in the upper 20 cm range were intensely contaminated by heavy metals due to the evaporation and capillary rise of the groundwater. The polluted groundwater from the Johannesburg mines was discharged into streams in the area and contributed up to 20% of the stream flow, causing an increase in the acidity of the stream water. The contamination of water from mines can

have far-reaching effects, extending over 10 kilometres beyond the immediate source. This contamination can pose significant environmental and health risks to surrounding ecosystems and communities (Naicker et al., 2003; Wade et al., 2002).

People and animals living in communities surrounding mine dumps are at greater risk of accessing contaminated surface water and groundwater. The build-up of contaminants in human bodies, exceeding allowable concentration limits, can lead to severe health effects. The World Health Organization (WHO, 2018) have described some of the by-products of the gold extraction process such as As, Cd, Hg, Mg, and Pb as possible human carcinogens of major concern health concern. These substances can be released into the environment during various stages of gold mining and processing, such as crushing, grinding, and smelting of ore (see also Kamunda et al., 2016).

In this study, boreholes at the Welkom campus of the Central University of Technology, Free State (hereafter CUT Welkom campus) were used as proxy to investigate and document the extent of contamination and groundwater vulnerability within the Welkom area. Through this study, a model will be developed to assess the extent of groundwater vulnerability to contamination from mine dumps. The groundwater vulnerability assessment method will be expected to assist and guide the local authorities, researchers, the public, and other interested stakeholders on the extent of groundwater contamination within the Welkom area and areas with similar environmental and geographical conditions.

1.2 Study rationale

People living in the communities surrounding the mine dumps in the Welkom area, including the students and staff at the CUT Welkom campus, are at a huge risk of exposure to contaminated groundwater. The continuous use and exposure of contaminated water may lead to the accumulation of contaminants in human bodies in concentrations that may lead to severe health effects, which may even result in death. It was thus paramount that this study accurately assessed and characterized the relationship between the contaminants in the mine dumps and those found in the groundwater sources around the Welkom area.

The effects of mine dumps on water quality have been broadly studied in many mining communities throughout the world. While many studies focused on how mining pollution

affects the surface water quality, very little attention has been given to the impact of mine dumps on groundwater quality. The major limitations to studying groundwater contamination in South Africa at scale is the availability of resources associated with the data complexities required by most vulnerability assessment techniques. Many groundwater assessment methods rely on simplified assumptions that may not fully capture the complexities of subsurface conditions, such as variations in geology, aquifer properties, water quality, as well as contaminant traveling and behaviours. In some cases, groundwater assessments tend to overlook the impacts of human activities such as land use changes, industrial operations, mining, and agricultural practices among others. Additionally, these methods may also struggle to adapt to local or site-specific conditions, thereby compromising their accuracy in predicting contamination risks. Lastly, the use of general indicators can result in oversimplifications, limiting the precision and reliability of the assessments.

This study attempts to remediate some of these gaps by developing a simple but efficient vulnerability assessment method that can be used to track and indicate the extent of groundwater contamination with simple resources. The outcomes of this study are expected to provide valuable insights for Welkom and other communities near mine dumps and groundwater sources, offering guidance on managing related environmental and health risks. These findings will benefit residents living close to gold mine dumps within the Lejweleputswa DM, as well as similar regions in South Africa and beyond. Furthermore, the study aims to inform local authorities in Welkom about the extent of environmental damage and health hazards caused by mine dumps in the area.

1.3 Research aims and objectives

This research aimed to investigate the ease with which contaminants from a mine dump can enter groundwater through the five boreholes located at the CUT Welkom campus, Lejweleputswa DM, in the Free State Province. The study aimed to achieve this through developing a new groundwater vulnerability model that addresses the specific vulnerability of the groundwater in the Welkom area relative to location and characteristics of the mine dumps. The new method was developed based on addressing the contextual factors that characterise specific vulnerability within the Welkom area, as well as on closing some of the methodological and procedural gaps and weaknesses

learned from previous methods and studies. Finally, a comparison was made between the outcomes of the new and existing groundwater vulnerability assessment methods.

These broad aims of the study were achieved through the following sub-aims and objectives:

SUB-AIM 1: ASSESSING THE GROUNDWATER VULNERABILITY TO CONTAMINATION FROM MINE DUMPS IN THE WELKOM AREA.

Objectives:

- To develop a new groundwater vulnerability assessment method that suits and applies to the context and characteristics of the Welkom area.
- To use the new groundwater vulnerability assessment method to assess the extent of groundwater vulnerability to pollution from mine dumps.

SUB-AIM 2: COMPARING AND VALIDATING THE EFFECTIVENESS OF THE NEW GROUNDWATER VULNERABILITY ASSESSMENT METHOD TO THE EXISTING VULNERABILITY ASSESSMENT METHODS.

Objectives:

- To apply some of the existing groundwater vulnerability assessment methods in assessing the groundwater vulnerability in the study area.
- To compare and validate the effectiveness of the new groundwater vulnerability assessment method with the existing methods.

SUB-AIM 3: USING THE OUTCOMES FROM THE GROUNDWATER VULNERABILITY ASSESSMENT TO MAKE RECOMMENDATIONS ON THE SAFETY AND USAGE OF THE GROUNDWATER IN THE WELKOM AREA.

Objective:

- To assess the environmental health implications of the Welkom groundwater as a result of the study findings.

All three sub-aims were deliberated on and broken down across the nine chapters.

1.4 Study hypothesis

The following null and alternative hypotheses were put forward in this study:

H₀₁ = The proximity of the mine dump to the boreholes located at the CUT Welkom campus has a significant contribution to the poor groundwater quality.

H₀₂ = The existing groundwater vulnerability assessment methods are not specific and therefore not effective in detecting the groundwater pollution from the mine dumps.

H₀₃ = The modified specific vulnerability assessment method is effective in detecting groundwater pollution in the Welkom area.

1.5 Definition of groundwater vulnerability

The term *groundwater vulnerability* emerged in the 1960s, primarily among European researchers and since then, groundwater vulnerability has been defined in many different terms. This is because the term *groundwater vulnerability* has been used in many different contexts and has thus assumed different meanings in time and application. In defining groundwater vulnerability, it is important to clarify and examine issues which relate to the “vulnerability of *what*, and vulnerability to *what*” (Harter, 2008).

According to Harter (2008) and Lobo-Ferreira and Oliveira (2003), the generally accepted definition of groundwater vulnerability is the sensitivity of groundwater quality to an imposed contaminant load, which is determined by natural characteristics or properties of an aquifer. Focazio et al. (2003) defined groundwater vulnerability as a measure of the likelihood of contamination to reach an aquifer. It is thus important to always note that groundwater vulnerability definitions can be referred to in many terms as some of the definitions are very contextual. Defining groundwater vulnerability definitions thus also helps in differentiating similar and associated vulnerability terms such as pollution risk, susceptibility risk, and contamination risk (Oke, 2015).

Table 1.1 shows some of the general definitions of vulnerability from some of the most used vulnerability terms:

Table 1.1 Definitions of groundwater vulnerability

No.	Definition	Sources *All these sources cited in Harter (2008)
1	“Vulnerability is an intrinsic property of a groundwater system that depends on the sensitivity of that system to human and/or natural impacts.”	International Association of Hydrogeologists (n.d.)

No.	Definition	Sources *All these sources cited in Harter (2008)
2	“Aquifer vulnerability is the possibility of ... diffusion of contaminants from the ground surface into natural water-table reservoirs, under natural conditions.”	Margat (1968), cited in Vrba and Zoporzec (1994)
3	“Vulnerability is the risk of chemical substances – used or disposed of on or near the ground surface – to influence groundwater quality.”	Villumsen et al. (1983), cited in Vrba and Zoporzec (1994)
4	“Groundwater vulnerability is the sensitivity of groundwater quality to anthropogenic activities which may prove [to be] detrimental to the present and/or intended usage-value of the resource.”	Bachmat and Collin (1987), cited in Vrba and Zoporzec (1994)
5	“Groundwater vulnerability is a measure of the risk placed upon the groundwater by human activities and the presence of contaminants ... Without the presence of contaminants, even the most susceptible groundwater is not at risk, and thus, is not vulnerable.”	Palmquist (1991), cited in Vrba and Zoporzec (1994)
6	“Groundwater vulnerability is the tendency of or likelihood for contaminants to reach a specified position in the groundwater system after introduction at some location above the uppermost aquifer”.	National Research Council, (1993), cited in Vrba and Zoporzec (1994)
7	“Groundwater vulnerability is defined as the tendency or likelihood of contaminants reaching the groundwater system after introduction at the surface and is based on the fundamental concept that some land areas are more vulnerable to groundwater contamination than others”.	Majandang and Sarapirome (2013)
8	“Vulnerability is the degree of endangerment, determined by natural conditions and independent of present source of pollution.”	Olmer and Rezac (1974), cited in Vrba and Zoporzec (1994).
9	Aquifer pollution vulnerability is about “the intrinsic character of the strata ... separating the saturated aquifer from the [immediately overlying] land surface” which determines the sensitivity to being adversely affected by a surface applied (anthropogenic) contaminated load.	Foster (1987)
10	“[Vulnerability of a hydrological system] is the ability of this system to cope with external, natural, and anthropogenic impacts that affect its state and character in time and space”.	Sotornikova and Vrba (1987), cited in Vrba and Zoporzec (1994)

As noted from these definitions, it is important to realise in principle that groundwater is always vulnerable to human activities. Groundwater vulnerability is high if natural factors provide little protection to groundwater, and groundwater vulnerability is low if the natural factors provide very little protection to the groundwater.

1.6 Research scope

The purpose of this study was to investigate the vulnerability of groundwater in Welkom from contamination by mine dumps. The study sought to develop a new groundwater vulnerability assessment method that applies to the context of the Welkom area and the conditions for contamination. The new groundwater vulnerability assessment method

includes parameters that can directly correlate with the vulnerability of the borehole water located at the CUT Welkom campus. The new groundwater vulnerability assessment method was tested and matched with the existing groundwater vulnerability assessment techniques.

The outcomes from the new specific groundwater vulnerability assessment method were then compared with the previous methods and validated to check on their effectiveness.

1.7 Structure of thesis

This thesis is divided into nine chapters which are structured as follows:

Chapter 1: *Introduction*

This chapter introduces the main purpose of the study, including its broad and specific objectives. It further provides the background to the study, the study rational, the aims and objectives for conducting the study, study hypothesis, definition of vulnerability terms, and the research scope.

Chapter 2: *Literature review*

This chapter provides a comprehensive literature overview on the general groundwater vulnerability definitions, methodologies of assessing specific vulnerabilities, and the application of specific vulnerabilities to point source pollution.

Chapter 3: *Physiography and geology of the Welkom area*

This chapter provides information on the Welkom and CUT Welkom campus study areas under investigation in this study. The chapter lays out the population, economic significance, as well as the climate and weather conditions in Welkom. The chapter also details the geology, stratigraphy, and types of soil found in the Welkom area.

Chapter 4: *Hydrological characterisation of the Welkom area*

This chapter examines and characterises the hydrology, hydrogeological, and geohydrological properties of the Welkom area under investigation. This is achieved by:

- assessing the water quality properties of the groundwater at the Welkom CUT Campus study site;
- assessing the Water Quality Index (WQI) methods;

- evaluating the WQI properties of the groundwater;
- characterisation of the aquifer systems in the study area; and
- evaluation and detection of aquifer fracture zones and groundwater flow direction.

Chapter 5: *Proposed groundwater specific vulnerability concept for abandoned mine dumps*

This chapter introduces the proposed groundwater specific vulnerability concept using the available data and knowledge shared from previous chapters. The new method is explained from its origins, data acquisition processes, application, and weaknesses.

Chapter 6: *Application of the DWAPH model to assess the impact of mine dumps on groundwater vulnerability in the Welkom area*

This chapter presents the applications of the new groundwater vulnerability assessment method named Direction of groundwater flow, Water quality, Aquifer type, Precipitation, Horizontal distance to contaminant source (DWAPH) in assessing groundwater vulnerability in the Welkom area.

Chapter 7: *Assessment of groundwater vulnerability in the Welkom area using other conventional groundwater vulnerability assessment methods*

In this chapter, the new groundwater vulnerability assessment method is compared with the established groundwater vulnerability assessment methods such as the Groundwater occurrence, Overlaying strata, depth to groundwater (GOD) method, aquifer vulnerability index (AVI), and the rainfall travel time (RTt) methods. Vulnerability maps from these methods are produced and compared to assess the effectiveness and efficiency of the new method.

Chapter 8: *Validation and significance of the DWAPH method*

This chapter presents the validation of the new groundwater vulnerability assessment method with nitrate (NO_3^-) concentration and microorganisms such as the total coliforms and *Escherichia coli* (*E. coli*). In this chapter, the similarities, differences, and correlations between the outcomes from the DWAPH method and the actual NO_3^- and bacterial concentrations are presented.

Chapter 9: *Conclusion, and recommendations*

This chapter integrates and summarises the outcomes and findings from this study with prior knowledge and suggests recommendations as well as possibilities for future research. The chapter concludes by highlighting the lessons learnt together with the gaps filled in by the research.

Chapter 2

Literature Review

2.1 Introduction

The concept of groundwater vulnerability assessment is very elusive (Khosravi et al., 2018; Moret, 2014). The vulnerability assessments referred to in many contexts speaks to the degree to which groundwater is exposed to intrinsic (natural) contaminants than specific contaminants (Anantha Rao et al., 2018; Linggasari et al., 2020; Putranto and Yusrizal, 2018). The groundwater vulnerability concept and its assessment was first introduced in 1968 by Margat, a French hydrogeologist, and since then, many more complex methods of groundwater vulnerability assessments have been developed throughout the world (Ewusi et al., 2017; Kumar et al., 2013; Kwon et al., 2022; Oke, 2020; Sakala et al., 2016). The definition of groundwater vulnerability is complex as various researchers reflected on various contextual definitions and methodologies for its assessment (Kwon et al., 2022; Misi et al., 2018; Ribeiro et al., 2017; Taghavi et al., 2023).

Groundwater vulnerability has been generally defined as the ability of the physical environment to resist the contamination of the groundwater from pollutants (Abdalla et al., 2010; Aller et al., 1987; Ducci and Sellerino, 2022; Margat, 1968; Oke et al., 2018). Natural groundwater vulnerability assessments are mainly based on the assumptions that the physical environment may provide some degree of protection to groundwater against natural and human impacts, and that some ground areas are more vulnerable to contamination than others, while specific vulnerability assessments focus on the effect of a particular pollutant or pollutants as the source of groundwater contamination (Abbasi et al., 2013; Oke et al., 2016; Sakala et al., 2016). While there seems to be an alignment between several groundwater vulnerability definitions, the methodological challenges pertaining to the measurement of groundwater vulnerability calculations varied largely due to the complexity of the contextual factors such as aquifer type, soil media, recharge, topography, and groundwater flow conditions (Bogomolov, 1966; Ducci and Sellerino, 2022; Kumar et al., 2013).

Groundwater vulnerability is a dimensionless, non-measurable, and relative property (Anantha Rao et al., 2018; Oke et al., 2016; Richts and Vrba, 2016; Wang et al., 2022).

This is mainly because, in most instances, groundwater vulnerability is made up by combining several key vulnerability characteristics that might be objectively or subjectively selected into one vulnerability equation or index. This combination may involve putting together various steps of selection, rating, and weighting of various dimensionless parameters using formulas across different measurement units (Chakraborty and Kumar, 2016; Khosravi et al., 2018; Mester et al., 2022; Sukmawati Rukmana et al., 2020; Vu et al., 2019).

Groundwater vulnerability assessment methods can be further classified into resource or source protection (Civita, 2010). While the former is primarily concerned with ensuring that sufficient quantities and good quality water are available throughout its surface manifestations, pulling it down via various geological strata, focus on the latter case is given to maintaining certain individual aquifers through horizontal movement (Oke, 2015).

This research sought to explore the various ways in which groundwater vulnerability assessment techniques can be utilised in assessing specific vulnerability to contamination across different contexts. This is because not much work has been invested into perfecting the techniques for specific groundwater vulnerability assessment methodologies. By drawing comparisons and deductions between studies conducted by various researchers on groundwater vulnerability assessment, this thesis sought to review and provide learning on closing some of the gaps, assumptions, and limitations that lie between various methodologies on intrinsic and specific groundwater vulnerabilities.

2.2 Intrinsic and specific groundwater vulnerability

Assessments of the vulnerability of groundwater can be divided into varieties of intrinsic and specific vulnerabilities (Falowo et al., 2017; Kogovšek and Petrič, 2013; Oke et al., 2018; Putranto and Yusrizal, 2018; Richts and Vrba, 2016). The concept of intrinsic vulnerability is based on the assessment of the effect of natural conditions such as climate, geology, and hydrological characteristics on groundwater contamination. Groundwater vulnerability is characterized as high if very little protection is provided by natural factors as they safeguard against contaminants at ground level, conversely, groundwater vulnerability is considered low, when natural factors offer relatively good protection against contaminants infiltrating the groundwater (Abbasi et al., 2013;

Ochieng et al., 2010; Putranto and Yusrizal, 2018). On the other hand, the concept of specific vulnerability refers to the effect of a particular contaminant, or a set of contaminants, on groundwater vulnerability (Anantha Rao et al., 2018; Ewusi et al., 2017; Falowo et al., 2017; Kumar et al., 2013; Kurwadkar, 2017). The degree or extent of specific groundwater vulnerability is mainly confined to the defined pollutant.

Both intrinsic and specific vulnerabilities are affected by processes and factors such as infiltration, recharge, depth to water table, soil type, rock type, retardation, and degradation of pollutants (Albinet and Margat, 1970; Benacchio and Civita, 1988; Foster, 1987; Linggasari et al., 2020). However, in some instances, contaminants are released from point sources where they infiltrate into the soil and percolate downwards through the unsaturated zone which neutralises the role of other factors including that of soil as an attenuation medium (Civita, 2010). It is thus sometimes possible to prepare risk scenarios that predict a specific vulnerability by considering the type of pollutant, their properties, and how they interact with the physical and chemical environment (Misi et al., 2018; Oke, 2020; Taghavi et al., 2023).

Field studies have demonstrated that specific vulnerability methods, which take into account factors such as land cover types and associated anthropogenic activities such as mining or agriculture, perform better at assessing groundwater contamination than intrinsic methods (McCarthy, 2011; Ribeiro et al., 2017). This is mainly because specific groundwater vulnerability assessment techniques can be employed to solve challenges from suspected anthropogenic pollutant sources.

2.2.1 Developing methodologies for assessing specific groundwater vulnerabilities

Groundwater contamination through anthropogenic activities has become a significant concern globally in the 21st century (Abdalla et al., 2010; Anantha Rao et al., 2018; Khosravi et al., 2018; Niedbalska, 2022; Richts and Vrba, 2016). The anthropogenic activities range from the poor management of mining waste or dumps, improper storage, and disposal of sewage wastewater by municipalities, poor management of leachate from municipal waste disposal sites, and the intensive use of pesticides in agriculture (Abdalla et al., 2010; Brahmi et al., 2021; Chakraborty and Kumar, 2016; Cossu et al., 2018; Liang et al., 2019; Linggasari et al., 2020; Mester et al., 2022; Vu et al., 2019).

The increase in human activities in modern times has resulted in the profound modification of the existing intrinsic groundwater vulnerability assessment techniques in efforts to trace the fate of various specific pollutants and their effect on the environment. This is mainly because specific vulnerability assessment techniques consider the link between pollutant type, exposure time, chemical nature, as well as the characteristics of the pollution source on the surface (Kogovšek and Petrič, 2013; Vu et al., 2019). As the factors affecting groundwater vulnerability, such as depth to water table, soil type, and recharge, are constantly changing due to the increase in anthropogenic influence, the concept of specific vulnerability assessment is becoming more broadly popular and relevant as compared to intrinsic vulnerability assessment (Richits and Vrba, 2016).

Most specific groundwater vulnerability assessment methods have been developed as modifications of intrinsic vulnerability assessment techniques (Neukum and Azzam, 2009). Factors such as the aim of the study, physiography of the tested areas, as well as the quality and quantity of the available data, influenced the extent of the modification of the groundwater vulnerability assessment techniques. Civita (2010) highlighted the division of groundwater vulnerability assessment techniques into two classes, namely for universal or local application situations. These two classes can be further subdivided into three basic groups:

- Homogeneous area zoning: Hydrogeologic complex and setting (HCS) assessment.
- Parametric system assessment: matrix systems, rating systems, point count system models.
- Analogical relation and numerical model assessment.

Of these three classes, the parametric assessment system is the most used due to its ease of application and usability in many scenarios (Abbasi et al., 2013; Brahmi et al., 2021; Cossu et al., 2018). Table 2.1 shows the various intrinsic vulnerability methodologies and their characteristics that can be modified or adapted for use in universal and or local applications.

Table 2.1 Methods of assessing aquifer vulnerability to pollution and the relative basic information

Soil and hydrological characteristics														References (All these sources are cited in Civita, 2010)
Type	Precipitation rate and chemical composition	Topographic surface and slope variability	Surficial stream-flow and network density	Thickness, texture, and mineralogy	Effective moisture	Permeability	Physical and chemical properties	Aquifer connections to surficial waters	Net recharge	Hydrogeologic features of ins zone	Depth to water	Aquifer hydrogeologic features	Aquifer hydraulic conductivity	
Hydrogeologic complex and setting								*		*	*	*	*	Albinet and Margat (1970), BRGM (1970)
Hydrogeologic complex and setting										*		*		Vrana (1968), Olmer and Rezac (1974)
Rating system				*					*	*	*	*	*	Fenge (1976)
Hydrogeologic complex and setting									*	*	*	*	*	Josopait and Swerdfeger (1976)
Rating system				*						*	*	*	*	Villumsen, Jaconsen and Sonderskov (1983)
Matrix system										*	*			Haertle (1983)
Hydrogeologic complex and setting	*			*						*		*		Vrana (1984)
Hydrogeologic complex and setting								*		*	*		*	Subirana, Asturias and Casas Ponsati (1984)
Matrix system								*		*	*	*		Engelen (1985)
Rating system				*	*	*	*			*	*	*		Zaporozec (edit.1985)
Hydrogeologic complex and setting				*	*	*	*	*	*	*	*		*	Breeuwsmas et al. (1986)
Rating system					*						*		*	Sotomikova and Vrba (1987)
Hydrogeologic complex and setting				*		*					*			Ostry et al. (1987)

Soil and hydrological characteristics														References (All these sources are cited in Civita, 2010)
Type	Precipitation rate and chemical composition	Topographic surface and slope variability	Surficial stream-flow and network density	Thickness, texture, and mineralogy	Effective moisture	Permeability	Physical and chemical properties	Aquifer connections to surficial waters	Net recharge	Hydrogeologic features of ins zone	Depth to water	Aquifer hydrogeologic features	Aquifer hydraulic conductivity	
Matrix system				*							*		*	Ministry Flemish Comm (1986); Goossens and Van Damme (1987)
Matrix system				*	*	*						*		Carter et al. (1987); Palmer (1988)
Rating system				*				*	*	*	*			Marcolongo and Pretto (1987) method 1
Analogical elation					*			*	*	*				Marcolongo and Pretto (1987) method 2
Rating system										*	*		*	GOD (Foster 1987, 1988)
Rating system				*			*		*	*				Schmidt (1987)
Point count system models	*	*			*			*	*	*			*	Troyan and Perry (1988)
Hydrogeologic complex and setting								*		*	*		*	GNDCl BASIC (Civita, 1990)
Point count system models		*		*						*	*	*	*	DRASTIC (Aller et al. 1985-1987)
Point count system models		*	*	*		*	*	*	*	*	*	*	*	SINTACS (Civita, 1991; Civita and De Maio, 1997, 2000)

Key: * Refers to the characteristics/parameters used by a particular method; GNDCl BASIC = National Group for the Defence against Hydrogeologic Disasters of the Italian National Council of Research ; DRASTIC = Depth to groundwater, Recharge, Aquifer type, Soil texture, Topography, Impact of vadose zone, and hydraulic Conductivity; SINTACS – Water table depth (S), Effective infiltration (I), Unsaturated zone (N), Soil media (T), Aquifer media (A), Hydraulic conductivity zone (C), and Topographic slope (S).

Like Civita (2010), several authors have also classified groundwater vulnerability assessment methods under three broad areas such as the statistical, the process-based and the index methods (Jhariya et al., 2016; Taghavi et al., 2023; Vu et al., 2019). Each of these methods has its own merits and demerits depending on its applications, for instance, while the process-based methods have proven to be more reliable in situations of estimating the migration of contaminants, they can be severely restricted by data shortages and computational difficulties (Liang et al., 2019). The choice of using a particular method thus heavily relies on the purpose and context of the study.

2.2.2 Homogeneous area zoning: Hydrogeological complex and setting methods

The HCS methods mainly consider the hydrogeology, geology, and the topographical maps above the lithology of an area (Musekiwa and Majola, 2013; Shah et al., 2021). The methods rely on the notion that two areas with similar characteristics in terms of hydrogeology are classified with the same groundwater vulnerability (Vrba and Zaporozec, 1994). The HCS method was first used by Margat (1968) and Albinet and Margat (1970) and is mainly applicable to small-scale mapping. The major benefit of using the HCS technique is that it allows for the identification of areas with substantial variations in geological formations such as in Karst; however, the validation of the HCS method in many areas remains a challenge (Ait Bahammou et al., 2019; Machiwal et al., 2018).

2.2.3 Parametric system assessment methods or overlay and index methods

The parametric system assessment methods are the most common approach in groundwater vulnerability mapping (Ducci and Sellerino, 2022; Moret, 2014; Sukmawati Rukmana et al., 2020). The groundwater assessment methods in the parametric system involve the identification and selection of factors or variables that are perceived to be critical in the formulation of any groundwater vulnerability assessment in any context.

The point count system, the rating system and the matrix system models all weigh in on the critical factors affecting groundwater vulnerability through activities happening in the vadose zone (Oke et al., 2018; Richts and Vrba, 2016). Each of the factors in the parametric system methods is assigned values in a spectrum range depicting various levels of contamination. These methods provide systematic frameworks for integrating

and analysing multiple factors that contribute to groundwater vulnerability. By quantifying vulnerability through matrix systems, rating systems, or point count system models, decision-makers can prioritize areas for management interventions, allocate resources effectively, and develop targeted strategies to protect and manage groundwater resources (Rahman, 2008; Saidi et al., 2011).

Parametric methods are built on the assumptions that a limited number of major parameters or factors predominantly influence groundwater vulnerability (Ribeiro et al., 2017; Vu et al., 2019). This assumption further builds on the principle that these factors are known or can be investigated and weighted. Parametric methods are thus mainly named based on the acronyms of factors from which they are developed. Some of the common parametric methods are DRASTIC (Aller et al., 1987), GOD (Foster, 1987), SINTACS (Civita and De Maio, 2000), EPIK (Epikarst, Protective cover, Infiltration conditions, and Karst network development) (Doerfliger and Zwahlen, 1998). Table 2.2 shows some of the advantages and disadvantages of the popular parametric methods, the RTAVI, GOD, and DRASTIC.

Table 2.2 Comparison of the AVI, GOD, and DRASTIC methods

Methods	Advantages	Disadvantages
Aquifer Vulnerability Index (AVI)	<ul style="list-style-type: none"> • Easy to apply. • Physical calculations conducted to obtain vulnerability level. • Uses only well log data. • Demonstrates a superior hydraulic resistance change interval. 	<ul style="list-style-type: none"> • Does not consider hydrogeological conditions. • The accuracy is not reliable. • Uses a few parameters and hence accuracy is not consistent.
GOD (Groundwater occurrence, Overlaying strata, Depth to groundwater)	<ul style="list-style-type: none"> • Easy to apply over wide areas with limited data. • Can distinguish between confined and unconfined aquifers as compared to AVI. 	<ul style="list-style-type: none"> • Depends on well log data and electrical sounding to understand the sub-surface conditions. • Subjective weighting system as it is obtained. • Less accurate as it considers less parameters such as in DRASTIC.
DRASTIC (Depth to groundwater, Recharge, Aquifer type, Soil texture, Topography, Impact of vadose zone, and Hydraulic conductivity)	<ul style="list-style-type: none"> • Have better accuracy as it considers more parameters. • Can be easily modified and adapted to context. 	<ul style="list-style-type: none"> • Have many parameters, which make it difficult to apply with limited resources. • Some parameters overlap and hence becomes redundant. • Weighting of some parameters is duplicated or not very influential.

		<ul style="list-style-type: none"> • Weighting of some parameters is subjective.
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Source: Putranto and Yusrizal (2018).

2.2.4 Analogical relation and numerical model assessment

The assessment methods within this category are relatively complex and involve the application of mathematical equations to represent the coupled processes governing contaminant transport. The analytical or numerical solutions detailed in the equations vary in complexity depending on the level of detail needed for the assessment (Kroeze et al., 2016; Ochieng et al., 2010). Process-based simulation models provide a rigorous way of understanding how easily groundwater can get contaminated by taking into account the many different factors that interact to affect contaminant movement.

Groundwater vulnerability maps that are set by analogical relation and numerical model assessments assist in setting priorities for identifying areas that are at greater risks of contamination by pollutants (Ali, 2007). Analogical and numerical assessments can be used to predict contamination movements in saturated and unsaturated zones of the vadose zone. This empowers local authorities from government departments and agencies to save time and funds in groundwater monitoring and protection. Most numerical methods such as SINTACS and the Modified DRASTIC can be applied in specific vulnerability mapping from medium to large scale areas (Anantha Rao et al., 2018; Ewusi et al., 2017). These models are favourable as they can target groundwater vulnerability from specific point pollution sources such as pesticides or NO_3^- . The analogical relation and numerical model assessment methods have the advantage that they can be easily applied and verified.

2.2.5 Statistical methods

Historically, statistical methods have not received as much attention as other vulnerability methods, however, they are increasingly being used to conduct vulnerability assessments of groundwater when some hydrogeological data is lacking (Liang et al., 2019; Niedbalska, 2022). Although statistical methods have high accuracy, they are less emphasised by experts for a variety of reasons. Some of these reasons include the complexity of implementing these methods, the need for specialised expertise, the availability of simpler and more accessible statistical approaches, and the historical

preference for traditional statistical methods. Additionally, the interpretation and communication of results from advanced statistical methods may be more challenging. However, it is important to note that there is an increasing appreciation of the significance of advanced statistical methods in addressing complex problems and making more accurate predictions.

The use of statistical methods can be an important factor in quantifying, determining, and evaluating the correlation between factors affecting groundwater vulnerability and the various measures of groundwater vulnerability. Statistical methods are formulated around the principle of uncertainty which can be linked to the distribution of probabilities of the groundwater parameters or variables of interest.

Ideally, statistical methods should be dominating other methods in the assessment of groundwater vulnerability. This is because groundwater vulnerability is a term that defines probability in its essence (Vu et al., 2019). Furthermore, statistical methods are flexible and easily adaptable for application at different scales as opposed to methods that are more inclined to the assessment of physical, and in some instances, chemical relationships between variables. Another advantage of using statistical methods is that they can be applied to different types of datasets, which can be qualitative, quantitative, or mixed methods (Kurwadkar, 2017). Variables from process-based methods such as Travel Time can be used for statistical analysis.

Despite the advantages of using statistical methods over other groundwater vulnerability assessment methods, statistical methods may not be effective in assessing groundwater vulnerability due to specific contaminants.

2.2.6 Process-based methods

Process-based models are mainly used for investigating groundwater vulnerability from a quantitative perspective as well for establishing a criterion for validating and comparing the performance of different models (Lindström, 2005; Putranto et al., 2018). In recent years, process-based models have been employed in site-specific studies, as well as in regional studies, and their uniqueness relies on their ability to predict the movement and transport of contaminants in both space and time. Process-based simulation models can assess the impact of biological, chemical, and physical processes on contaminant transport. This includes the effects of microbial activity, chemical reactions, and

physical properties of the subsurface on the contaminant fate and transportation (Richts and Vrba, 2016).

Some of the advantages of applying process-based methods are:

- They can be applied to many contexts as they attempt to analyse the biological, chemical, and physical reactions that happen from the surface through to the groundwater.
- They can be used to study processes in hydrogeological settings that are generic, and can be useful for defining regulatory guidelines and predictions for hazards at specific sites.

However, process-based methods, like any other groundwater vulnerability assessment method, have their own limitations. The major limitations to applying process-based models are largely due to their formulation as they entail a lot of mathematical processes. Moreover, the need for extensive quality data inputs can be a limitation in maintaining the efficiency of the method (Kroeze et al., 2016).

The general observation is that the methods become more efficient as they include more parameters. However, the downside of having more parameters is that the method becomes more complex to apply and would require more resources to operate. The development of groundwater vulnerability assessment methods should thus seek to create a balance between improving efficiency and managing operational complexities that comes with the new method.

2.3 Application of specific groundwater vulnerability assessment methods

2.3.1 DRASTIC method

At any instance, the criteria for the selection of a particular groundwater vulnerability assessment method may vary depending on the availability or paucity of data, as well as the geological nature of the study area. Among the most popular and widely used methods for assessing groundwater specific vulnerability is the DRASTIC method. The DRASTIC method is an overlay-index method developed by Aller et al. (1987). It uses seven hydrogeological parameters that control the movement of contaminants in an aquifer (Abbasi et al., 2013; Khosravi et al., 2018; Liang et al., 2019; Taghavi et al., 2023).

DRASTIC is an acronym that stands for the hydrologic features of Depth to groundwater, Recharge, Aquifer type, Soil texture, Topography, Impact of vadose zone, and hydraulic Conductivity. The overall DRASTIC index is a numerical result that is formed from the ratings and weights assigned to each of the seven parameters based on their contamination probabilities.

The original DRASTIC equation is shown in Equation 2.1:

$$DRASTIC\ index = DrDw + RrRw + ArAw + SrSw + TrTw + Irlw + CrCw \quad \text{Equation 2.1}$$

Where:

- D = the depth to water level
- R = the recharge
- A = Aquifer type
- S = the soil structure
- I = the impact of the vadose zone
- T = the topography
- C = hydraulic conductivity
- r* = the rating of the parameter
- w* = the weight of the parameter.

The popularity of the DRASTIC method stems from its flexibility and ability to be combined with other parameters such as land use, aquifer thickness, lineament, impact of contaminant and its adaptability to the confined hydrogeological conditions and data availability in specific areas. Figure 2.1 shows the diagram for the DRASTIC method and its characteristics.

Many researchers have used the addition or removal of parameters from the original DRASTIC method to modify its function from assessing intrinsic to specific vulnerability. The addition of parameters such as land use may possibly affect the groundwater recharge process and plays an important role in the evaluation of the effects of anthropogenic activities on groundwater quality (Misi et al., 2018; Rodriguez-Galiano et al., 2014).

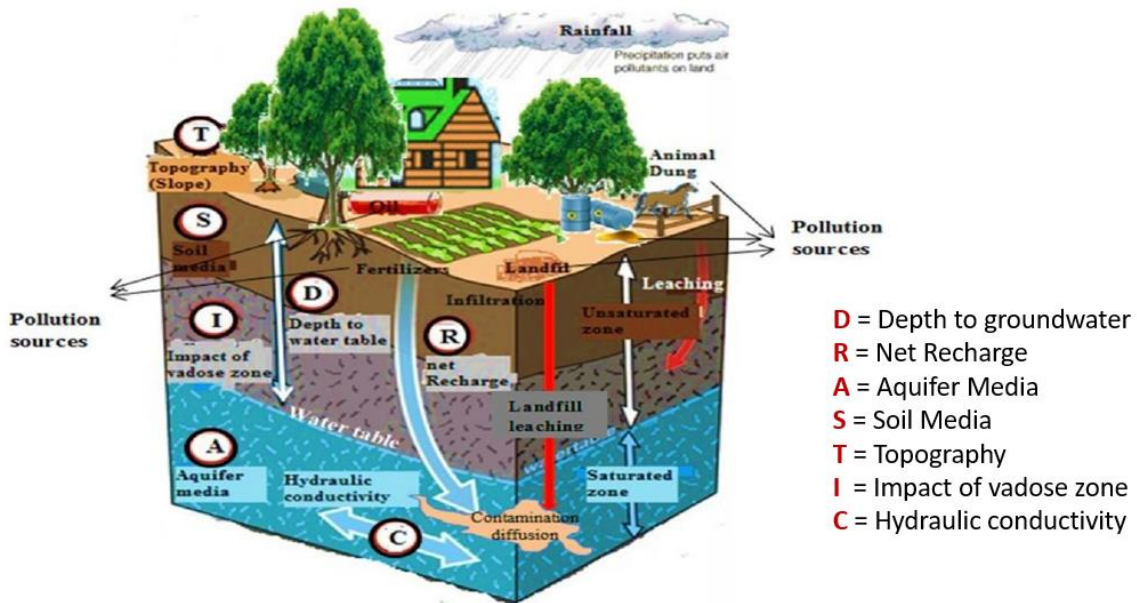


Figure 2.1 DRASTIC method and its parameters (Greater Oliver Groundwater Vulnerability, n.d.)

Despite its popularity, the original DRASTIC model was designed without a feed validation mechanism of tracing a particular pollutant or set of pollutants such as the measurement of NO_3^- concentrations in groundwater. In addition, the weights of the factors in the DRASTIC model were allocated based on the original pre-set values proposed by Aller et al. (1987), and the rates or sub-factors were chosen based on experts' judgement that introduced human subjectivity, uncertainty, and error. Because of this, various researchers have put forward numerous modified versions of the intrinsic DRASTIC method to remediate these shortfalls. The addition or removal of additional factors such as irrigation type and land-use are the most common approaches to modifying the DRASTIC method, however, other scholars have adapted the DRASTIC method based on the hazard maps and weight and rate scores on field measurement data (Anantha Rao et al., 2018; Khosravi et al., 2018; Liang et al., 2019; Taghavi et al., 2023).

2.3.2 Application of the modified DRASTIC method in assessing mine dump pollution

Mining activities can cause devastating changes to the environment and groundwater resources through changes in the morphology and geology of an area, land use, and the accidental or intentional disposal of solid or liquid waste or by-products from the mining processes (Abdalla et al., 2010; Ewusi et al., 2017; Linggasari et al., 2020; Sakala et al.,

2016; Sukmawati Rukmana et al., 2020). Groundwater in mining areas is known to be vulnerable to quality problems that may have severe impacts on human health. The contamination of groundwater in mining areas through leakage and spillage of chemicals from mining activities such as cyanide and mercury result in it becoming unsafe for human consumption (Ewusi et al., 2017; Ochieng et al., 2010). It is thus crucial to regularly assess the vulnerability of groundwater in areas near mining sites or activities.

Several groundwater vulnerability assessment methods have been utilised in many contexts to evaluate the ease of specific vulnerability of groundwater from mining activities. In most cases, rainwater draining the mine plateau flows and percolates into the shallow alluvium aquifers, thereby polluting the groundwater (Richs and Vrba, 2016).

Studies conducted by Linggasari et al. (2020) to assess the impact of coal mine dumps on the environment at the Tanah Laut district in South Kalimantan, utilised the modified DRASTIC method through adding the distance from lineament and land use parameters. The modified DRASTIC method made use of the analytic hierarchical process to enhance the weights and rates of the DRASTIC parameters and the resultant DRASTIC index was validated against eight chemical parameters: iron (Fe), manganese (Mn), chloride (Cl), sulphate (SO_4), calcium (Ca), magnesium (Mg), total suspended solid (TSS) and pH. Further assessment conducted on the correlation coefficient of these eight parameters with the DRASTIC index indicated that Fe had the greatest correlation coefficient value of 81%, compared to the other parameters. The advantage of validating the DRASTIC index against the eight physical and chemical parameters was that it provided the range of specificity in identifying and detailing the extent of groundwater vulnerability to contamination.

Figure 2.2 shows the overall and specific vulnerability maps due to Fe concentration in the Tanah Laut district. The overall study results revealed that close to 24% of the total study area lay in the high to very high vulnerability zones due to the effect of land degradation caused by coal mining on groundwater.

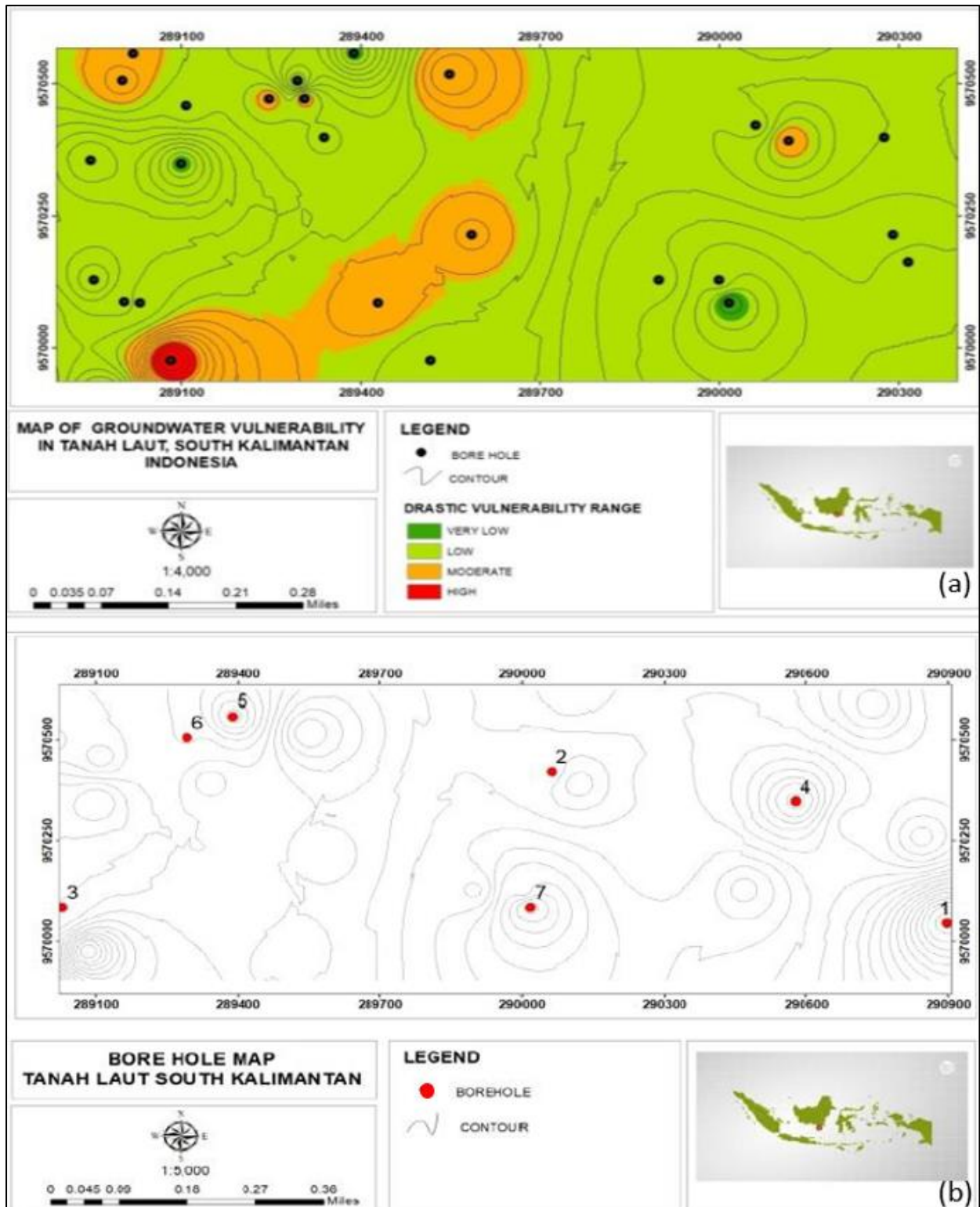


Figure 2.2 Groundwater vulnerability maps in Tanah Laut, South Kalimantan, Indonesia: (a) general vulnerability map, (b) specific vulnerability to iron concentration (Linggasari et al., 2020)

Sometimes the use of validation techniques alone against groundwater vulnerability assessment methods is not sufficient in determining the extent of groundwater vulnerability to contamination. This is because some of the parameters within groundwater vulnerability assessment methods may not be consistent with the contextual factors affecting groundwater vulnerability in some areas or regions. To improve the specificity and maximise the efficiency of some groundwater vulnerability assessment methods, some researchers have sought to altering, modifying, or removing some parameters from the original established groundwater vulnerability assessment methods. In their assessment of the effects of AMD on the groundwater vulnerability in the Witbank coalfield in South Africa, Sakala et al. (2016) modified the DRASTIC method to the DRIST method.

Unlike the usual DRASTIC method with seven parameters, the DRIST method was based on five parameters, namely Depth to water level, net Recharge, Impact of the vadose zone, Soil media and Topography. The parameters used in the DRIST index method were as follows:

$$DRIST\ index = D_r D_w + R_r R_w + I_r I_w + S_r S_w + T_r T_w \quad \text{Equation 2.2}$$

Where:

- D = depth to water level
- R = recharge
- I = impact of the vadose zone
- S = soil structure
- T = topography
- r = rating of the parameter
- w = weight of the parameter.

The aquifer media and hydraulic conductivity parameters present in the original DRASTIC method were not utilised in the DRIST method as they were difficult to estimate for the fractured aquifers that dominated the study area. Moreover, the DRIST method only dealt with the subsurface conditions before pollutants enter the aquifer, thereby assuming that the aquifer media and hydraulic conductivity has little to no impact on groundwater vulnerability to contamination. An artificial neural networks (ANN) model was built using the DRIST input parameters together with the hydrochemistry data comprising of SO_4 and

total dissolved solids (TDS) as training samples to produce the groundwater assessment model of the Witbank coalfield. The result of the DRIST-ANN model correlated perfectly with the physically measured pH values. The DRIST-ANN model could thus be relied upon in assessing the groundwater vulnerability that generates SO_4 in fractured aquifers in previous coal mining communities (Sakala et al., 2016). However, despite its overall success in determining the extent of groundwater vulnerability to contamination in the Witbank area, the application of the DRIST-ANN model may be limited in other areas, as some of the parameters may not correlate with the physical, chemical, and hydrological parameters in other areas.

Figure 2.3 shows the DRIST input parameters on a Geographical Information Systems (GIS) map of the Witbank coalfield as suggested by Sakala et al. (2016).

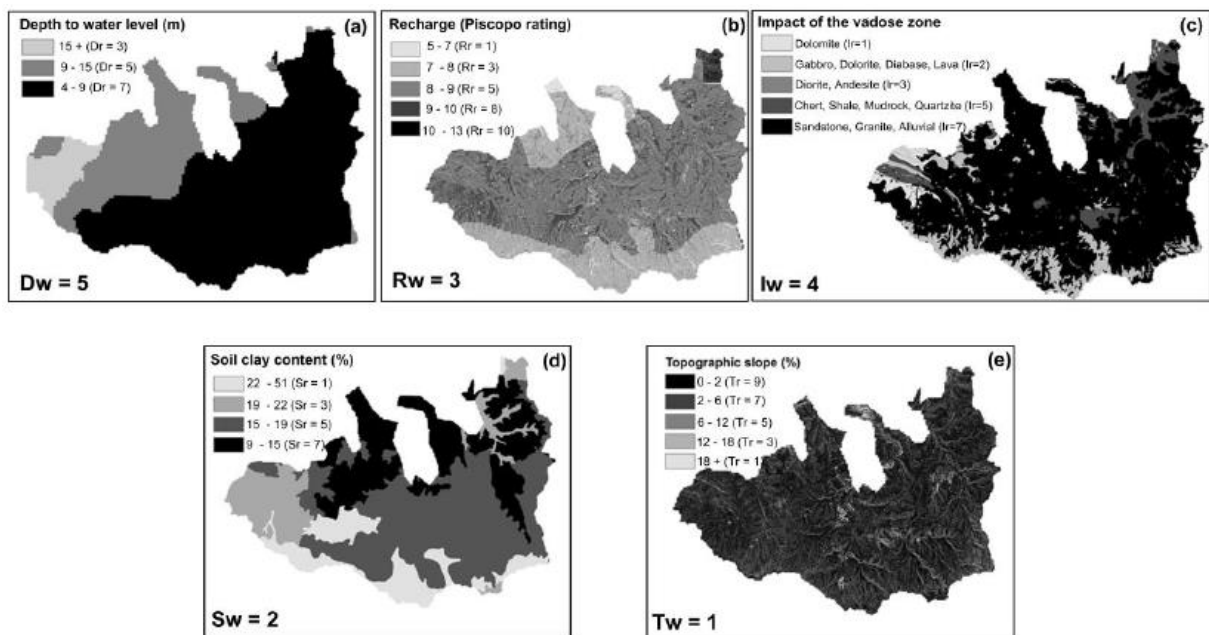


Figure 2.3 DRIST inputs : (a) Depth to water level; (b) Recharge; (c) Impact of the vadose zone; (d) Soils; and (e) Topography (Sakala et al., 2016)

2.3.3 Application of the pit groundwater vulnerability to pollution index method in assessing mine dump pollution

In a rare case of groundwater contamination, the use or transfer of post-mining residue from mines to other areas can be a cause of secondary groundwater contamination. The development of the modified parametric groundwater vulnerability assessment was carried out for an open pit mine exploiting quaternary filling sand in southern Poland.

Niedbalska (2022) utilised the modified rank–weight method of evaluating the groundwater vulnerability and adapted it to the atypical type of a pollutant emitter. The value index for the groundwater vulnerability to pollution was named as the Pit Groundwater Vulnerability to Pollution Index (PGVPI), which was calculated as follows:

$$PGVPI = [Ta(w).Ta(r)] + [Ts(w). Ts(r)] + [R(w). R(r)] + [M(w). M(r)] + [Pa(w). Pa(r)] + [Ps(w). Ps(r)] \quad \text{Equation 2.3}$$

Where:

PGVPI = pit groundwater vulnerability to pollution index

Ta = thickness of the waste in the aeration zone

Ts = thickness of the waste in the saturation zone

R = aquifer recharge from precipitation

M = terrain topography

Pa = filtration coefficient of the formations building the aeration zone

Ps = filtration coefficient of formations building the saturation zone

w and r are the lower indices denoting the weight and rank of the parameter, respectively.

The selection criteria for these factors were the evaluation of the relationship between the dump and the groundwater table. One of the main assumptions in the proposed method was the effect of the groundwater on the least bottom of the collected waste pit dumps. Table 2.3 shows the parameters with the rank and weights of the Pit Groundwater Vulnerability Index (PGVI) method.

Table 2.3 Parameters with rank and weights of the PGVI method (Niedbalska, 2022)

Factor/parameter	Parameter value	Rank	Weight
T _a waste thickness in the aeration zone [m]	0.0 ÷ 2.0	3	5
	2.0 ÷ 7.0	5	
	7.0 ÷ 13.0	7	
	13.0 ÷ 18.0	8	
	18.0 ÷ 25.0	9	
	> 25.0	10	
T _s waste thickness in the saturation zone [m]	0.0 ÷ 4.0	3	6
	4.0 ÷ 9.0	5	
	9.0 ÷ 15.0	6	
	15.0 ÷ 22.0	8	
	22.0 ÷ 30.0	9	
	> 30.0	10	
R infiltration recharge [m/d]	0.0 ÷ 0.000089	1	3
	0.000089 ÷ 0.000178	3	
	0.000178 ÷ 0.000267	5	
	0.000267 ÷ 0.000356	6	
	0.000356 ÷ 0.000445	8	
	> 0.000445	10	
M land surface morphology [%]	0.0 ÷ 2.0	10	2
	2.0 ÷ 6.0	9	
	6.0 ÷ 12.0	5	
	12.0 ÷ 18.0	3	
	> 18.0	1	
P _a aeration zone filtration coefficient [m/d]	0.0 ÷ 0.0000001	1	5
	0.0000001 ÷ 0.000001	2	
	0.000001 ÷ 0.00001	4	
	0.00001 ÷ 0.001	6	
	0.001 ÷ 0.1	8	
	> 0.1	10	
P _s saturation zone filtration coefficient [m/d]	0.0 ÷ 0.0009	1	6
	0.0009 ÷ 0.09	2	
	0.09 ÷ 0.9	4	
	0.9 ÷ 9.0	6	
	9.0 ÷ 90.0	8	
	> 90.0	10	

2.3.4 Application of the modified GOD method in mine dump pollution

In their assessment of the groundwater vulnerability in a schist belt environment in south-western Nigeria, Akinlalu et al. (2021) modified the GOD vulnerability assessment method to the GODL, by adding the longitudinal conductance parameter. The assessment of the geophysical conditions of the study area indicated that Groundwater

confinement, aquifer Overlying strata, Depth to water table, and Longitudinal conductance were the main groundwater vulnerability causative factors. Figure 2.4 shows an illustration of the addition of the longitudinal conductance parameter to the GOD method.

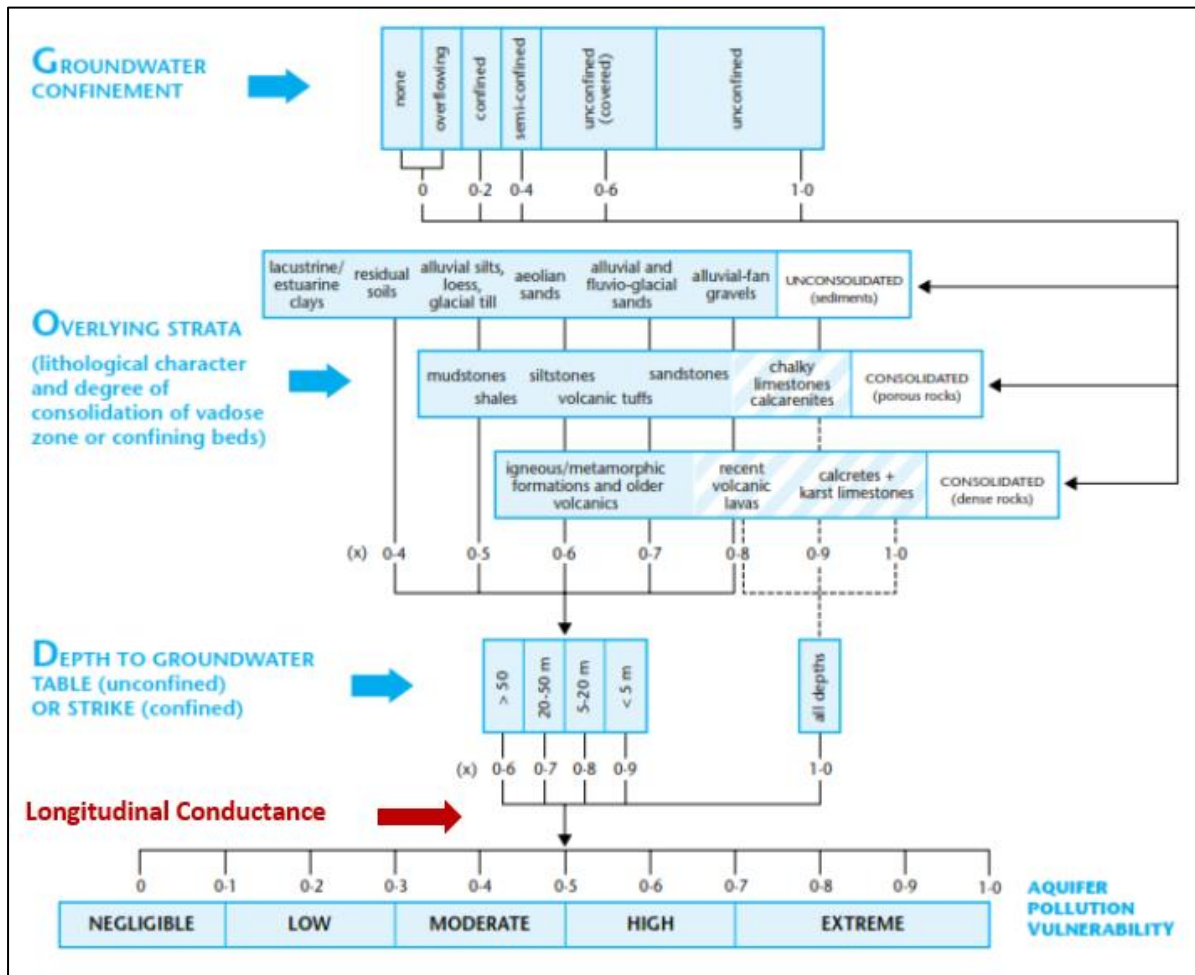


Figure 2.4 Addition of the longitudinal parameter from the original GOD to GODL method (Sukmawati Rukmana et al., 2020)

The aquifer vulnerability causative factors identified were inserted in the GIS by employing the conventional GOD and the analytic hierarchy process (AHP)-driven GODL algorithms. The validation of the results conducted by using water chemistry elements indicated that the GODL model demonstrated 70% accuracy, compared to the 66% accuracy level indicated by the original GOD model. This study further demonstrated that the contextualisation of groundwater vulnerability assessment methods contributes immensely to the efficiency of the groundwater vulnerability assessment outcomes. Contrary to Akinlalu et al.'s (2021) findings, studies carried out by Sukmawati Rukmana et al. (2020) found a very negligible to moderate impact of coal mining to groundwater in

their assessment of the impact of coal mining to groundwater vulnerability in Balangan, South Kalimantan, Indonesia, using the unmodified GOD method. This further proved the importance of adapting groundwater vulnerability assessment methods to solve groundwater-specific vulnerability challenges.

2.3.5 Application of the susceptibility index method in detecting nitrate contamination

As in mining, conventional groundwater vulnerability assessment methods can be modified to assess the specific vulnerability of groundwater to NO_3^- (Vu et al., 2019). In recent studies, the addition of the land use parameter to general groundwater vulnerability assessment methods has helped improve the efficiency of the groundwater vulnerability maps by detecting the impact of human activities such as agriculture on groundwater quality (Khosravi et al., 2018; Liang et al., 2019).

Ribeiro et al. (2017) applied the susceptibility index, a modification of the DRASTIC method, in the evaluation of groundwater vulnerability induced by rice cultivation in the Daule aquifer, Guayas River basin in Ecuador. The outcomes of the assessment indicated a high recharge and flat topography as the main contributors to high vulnerability. The inclusion of the land use index benefited calculations on the vulnerability assessment on diffuse agricultural pollution. The results obtained and the subsequent validation using an NO_3^- concentration, measured in field, indicated the advantage of the susceptibility index over the original DRASTIC, as the DRASTIC method revealed an underestimation of the vulnerability for almost the entire aquifer area. The incorporation of the new land use parameters as shown in Figure 2.5 allowed the inclusion of the human induced factors in the calculations, thus transforming the index from intrinsic to specific (Ribeiro et al., 2017).

Pouye et al. (2022) modified the DRASTIC method by adding NO_3^- in their investigation of vulnerability of the unconfined aquifers in the quaternary sands in Thiaroye, Dakar, Senegal.

Table 2.4 shows the different land use classes with their respective ratings as identified by Ribeiro et al. (2017).

Table 2.4 Land use classes and respective rates used in the susceptibility index method

Land use	Rating
Agricultural areas, irrigation perimeters (annual crops), paddy fields	90
Permanent crops (orchards, vineyards)	70
Heterogeneous agricultural areas	50
Pastures and agroforestry areas	50
Artificial areas, industrial waste discharges, landfills	100
Quarries, shipyards, open-air mines	80
Continuous urban areas, airports, harbours, (rail)roads, areas with industrial or commercial activity, laid out green spaces	75
Discontinuous urban areas	70
Natural areas, aquatic environments (salt marshes, salinas, intertidal zones)	50
Forests and semi-natural zones	0
Water bodies	0

Source: Ribeiro et al. (2017)

2.3.6 Application of the modified SINTACS method in detecting nitrate contamination

Kumar et al. (2013) applied the SINTACS model in assessing the specific groundwater vulnerability assessment of the Kancheepuram district, in Tamil Nadu, India, to generate a small-scale map of groundwater vulnerability to contamination.

The SINTACS model was derived from the DRASTIC model by Civita and De Maio in 1997 with seven parameters:

- S Water table depth
- I Effective infiltration
- N Unsaturated zone
- T Soil media
- A Aquifer media
- C Hydraulic conductivity zone
- S Topographic slope

The result showed a very low to moderate vulnerability to NO₃- pollution. The model was proven with groundwater quality data and the results indicated a very strong correlation between the SINTACS-specific vulnerability index and NO₃- concentrations as used for validation. Figure 2.5 shows the SINTACS methodology, indicating the validation by NO₃-.

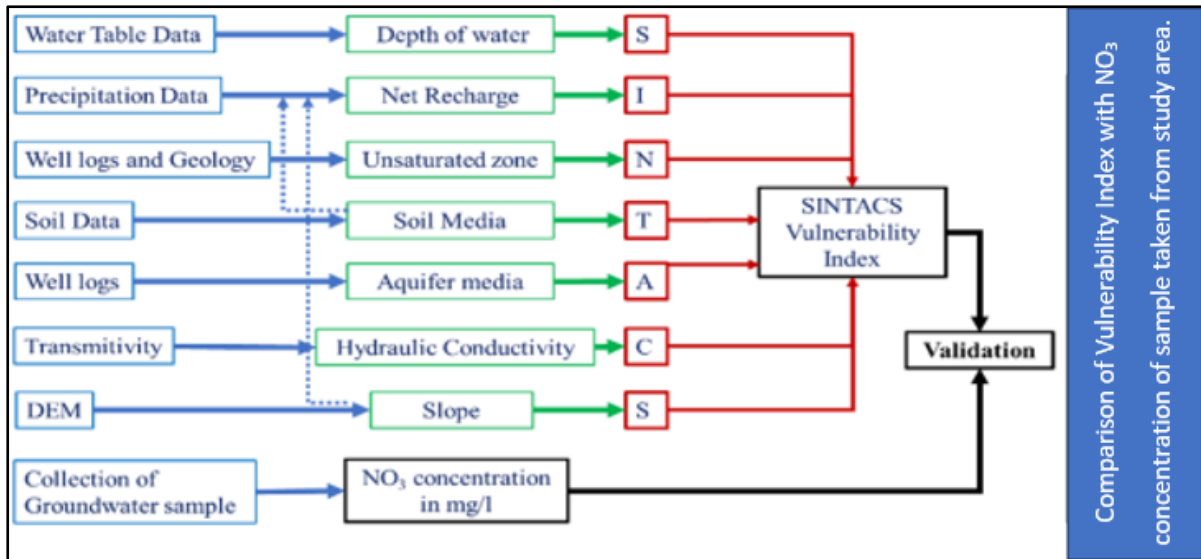


Figure 2.5 SINTACS vulnerability index methodology (Sahu et al., 2022)

Javadi et al. (2011) also applied the modified SINTACS method in identifying the groundwater vulnerability from NO_3^- in agriculture, settlement areas, and wastelands. Specific ranges were assigned different ratings depending on the potential of NO_3^- contamination from different sources. For each of the contaminants, an additional parameter was linearly combined with the SINTACS vulnerability index to calculate the specific SINTACS vulnerability index, which was computed using the following equation (Javadi et al., 2011; Al Kuisi et al., 2006):

$$SIVI = \sum_{i=1}^7 P_i * W_i \quad \text{Equation 2.4}$$

Where:

SIVI = SINTACS vulnerability index

P_i = the rating of each of the seven parameters that the method considers, and

W_i = the relative weight.

This index is mostly useful at a regional scale to priority areas of high, moderate, low, and very low vulnerability regions. The SINTACS specific vulnerability index was then calculated using the following equation:

$$SSVI = SIVI + AI_r * AI_w \quad \text{Equation 2.5}$$

Where:

SSVI = SINTACS specific vulnerability index

AI = anthropogenic parameter

- r = corresponding rating
- w = corresponding weight.

The model was modified by involving an additional parameter reflecting the anthropogenic impact.

In many instances, the vulnerability method for NO₃⁻ contamination can be used as a preventative measure. Based on the intrinsic vulnerability determined by the original DRASTIC method, Kwon et al. (2022) added the land use and hydraulic gradient parameters to assess the effects of NO₃⁻ contamination in the Jeju Volcanic Island in Korea. The resultant specific vulnerability index was assessed using GIS as shown in Figure 2.6.

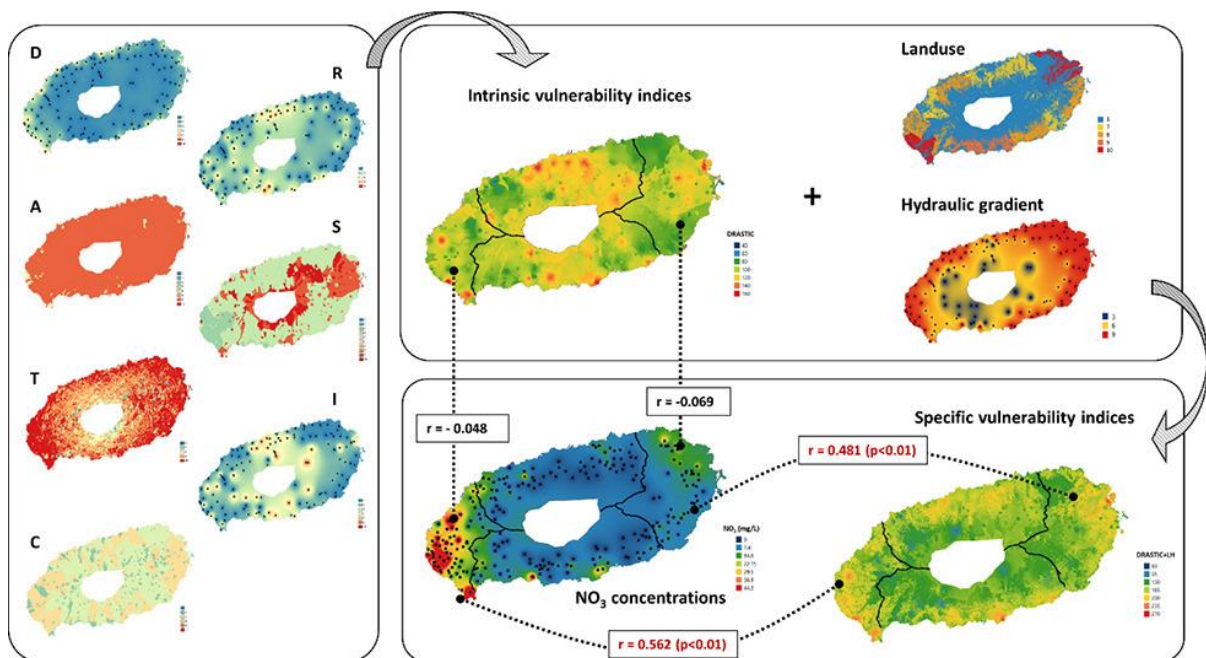


Figure 2.6 Assessing the nitrate vulnerability of groundwater in Jeju Volcanic Island, Korea (Kwon et al., 2022)

Sometimes the prediction accuracy of conventional groundwater vulnerability assessments, such as the DRASTIC method, is severely limited by the inherent subjectivity in determining the main parameters, rating scales, and weighting coefficients. In determining the groundwater vulnerability to NO₃⁻ contamination due to intensive fertiliser application in the Datong basin in the Shanxi Province in China, Liang et al. (2019) modified the conventional DRASTIC method to the DRSTIC-LE method. The DRSTIC-LE method was used to assess the specific vulnerability due to NO₃⁻, and

comprised of the Depth to water table, net Recharge, Soil media, Topography, Impact of the vadose zone, hydraulic Conductivity, Land use type, and groundwater Exploitation as evaluation parameters. The addition of the land use type and groundwater exploitation enhanced the sensitivity of the assessment according to the results of the groundwater-specific vulnerability map, which indicated a very high vulnerability in the central, northern, and north-eastern portion of study area of the Datong basin. Additionally, by comparing the correlation between the effective weights and the theoretical weights calculated by the entropy weight (EW), AHP and the Entropy weight–Analytic hierarchy process (EW–AHP) method, it was found that the prediction accuracy of the EW–AHP method was higher than other methods and hence effective for determining weights as a basis for regional control of groundwater NO_3^- pollution.

Taghavi et al. (2023) also compared the use of the DRASTIC and DRASTIC-L (land use) groundwater vulnerability assessments in relation to NO_3^- contamination due to the extensive use of fertilisers in the Burdekin Basin, Queensland, Australia, and found the DRASTIC-L method more effective than the DRASTIC method. A comparison of the DRASTIC groundwater vulnerability index and water quality parameters, including NO_3^- , indicated very low correlations, while the DRASTIC-L method resulted in a higher correlation with the water quality parameters including the NO_3^- concentration.

2.3.7 Application of physical methods in assessing groundwater vulnerability to leachate contamination

Several studies have focused on the vulnerability of groundwater to landfill leakage. The lack of proper management and treatment of waste has resulted in the contamination of groundwater resources in many parts of the world (Cossu et al., 2018; Kogovšek and Petrič, 2013; Li et al., 2022; Mester et al., 2022). Eliminating sources of pollution and protecting aquifers, has become a major challenge of the 21st century in most regions of the world and thus assessing the vulnerability of groundwater contamination due to liquid landfills is critical (Brahmi et al., 2021; Chakraborty and Kumar, 2016; Mosuro et al., 2017).

The contamination of groundwater from landfill leakage is usually characterised by high levels of dissolved organic matter, inorganic nitrogen, and phosphorous compounds. Ammonium (NH_4^+) is commonly present in municipal wastewater and landfill leachate at very high concentrations, compared to environmental quality standards. It is often the

key pollutant in environmental risk assessments of solid and liquid landfills, sewers, and contaminated sites due to its abundance and mobility (Li et al., 2022; Niedbalska, 2022). The oxidation of NH_4^+ results in the generation of nitrite (NO_2^-), which is converted into NO_3^- in the final stage of nitrification. NO_3^- is extremely mobile due to its high solubility in water and can thus be found in many groundwater sources (Wang et al., 2022).

Although rarely used due to their complications in calculations, physical methods have been commonly used in assessing the specific groundwater vulnerability through leachates. This is mainly because leachate plumes are often more electrically conductive than the surrounding pore waters and thus can be easily detected by an electrical geophysical method (Chakraborty and Kumar, 2016; Cossu et al., 2018; Kogovšek and Petrič, 2013). Brahmi et al. (2021) utilised the electrical resistivity and induced polarisation imaging survey to investigate the effects of leachate contamination of groundwater in the Tebessa basin in Algeria. In their investigation, Brahmi et al. (2021) used the geochemistry analysis based on atomic absorption spectrometry, and geophysics prospecting based on electrical resistivity tomography and induced polarisation techniques. The study results showed high levels of mineralisation consisting of Pb, Cd, Cu, and Zn in the vadose zone due to the leakage of leachate.

The electrical resistivity method is most frequently used in environmental studies because the electrical resistivity of earth materials is determined by parameters such as fluids, conductivity of the matrix, porosity, permeability, temperature, degree of fracturing, grain size, degree of cementation, rock type and the extent of weathering of the medium (Brahmi et al., 2021; Li et al., 2022). Mosuro et al. (2017) used the resistivity method to assess the groundwater vulnerability in Western Nigeria. They found the results from the electrical sounding correlating positively with those of electrical resistivity tomography, because zones of leachate contamination are characterised by very low resistivity values relative to the background resistivity of rocks (Misi et al., 2018).

According to Oke (2015), most physically-based methods require numerical solutions to processes that utilise contaminant transport. This has the advantages of managing huge datasets together with challenges of adjusting the data scales of results.

2.3.8 Application of the random forest regression method in predicting groundwater vulnerability to nitrate

Rodriguez-Galiano et al. (2014) conducted a study in Vega de Granada in Spain to evaluate the vulnerability of groundwater to NO_3^- pollution using the random forest (RF) regression model. The RF model is a learning technique that uses algorithms for classification and regression, which was developed by Breiman (2001). In their study, Rodriguez-Galiano et al. (2014) used the RF method to develop groundwater vulnerability maps with concentrations of NO_3^- above 50 mg/l in the Vega de Granada aquifer. The advantages of the data maps were that they can be used for identifying areas at risk of NO_3^- pollution and establishing monitoring programmes that can remediate the causes of the NO_3^- pollution in groundwater. Figure 2.7 shows the RF regression flowchart used for the training, sample, and test datasets.

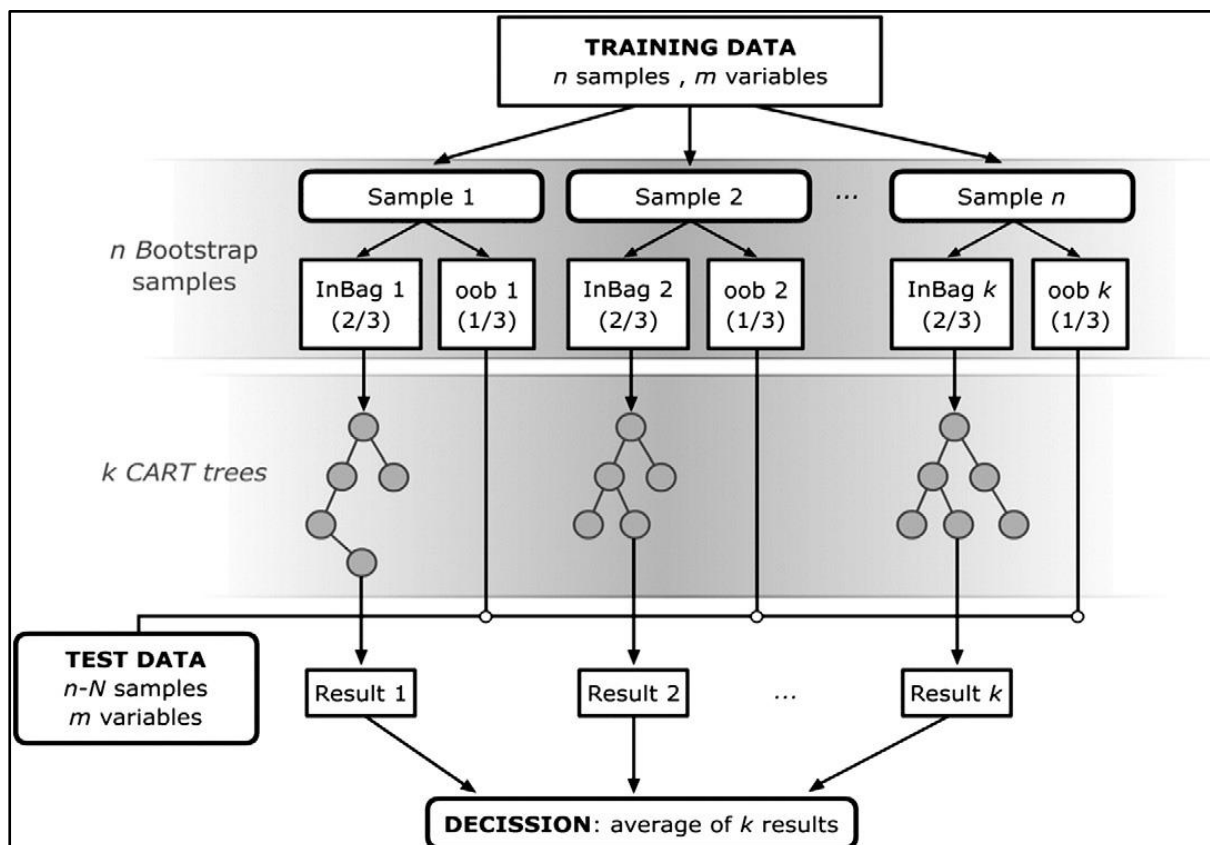


Figure 2.7 Random forest regression flowchart (Guo et al., 2011)

Although rarely used in the groundwater vulnerability assessments, the RF method has several advantages compared to conventional assessments methods:

- It is non-parametric.

- It has a high predictive efficiency and accuracy rate.
- It has the ability to determine variable importance that can be utilized to better comprehend the individual and combined effect of explanatory variables in both safeguarding and exposing groundwater from and to a pollutant.

Despite these advantages, the RF method requires complex mathematical understanding that may lead to errors, if not properly conducted.

2.3.9 Application of the rainfall travel time method

Using the RTt method, Oke (2015) was able to identify the high risk of groundwater vulnerability in the Dahomey Basin in south-western Nigeria. The RTt is a simplified groundwater vulnerability assessment concept that employs the source–pathway–receptor approach. The design of the model uses both subjective and objective criteria in the identification and use of parameters such as the travel time of contaminants to the vadose zone, recharge, slope, infiltration, thickness of the vadose zone and hydraulic conductivity. Figure 2.8 shows the objective and subjective criteria of the RTt method.

Compared to other groundwater assessment techniques, the RTt method has the following advantages:

- It can be applied for use in data scarce areas.
- It utilises very few parameters.
- It is comparatively efficient compared to other groundwater vulnerability techniques.

However, the major limitations for the RTt method are that it does not account for anthropogenic activities, and it does not consider contaminant types.

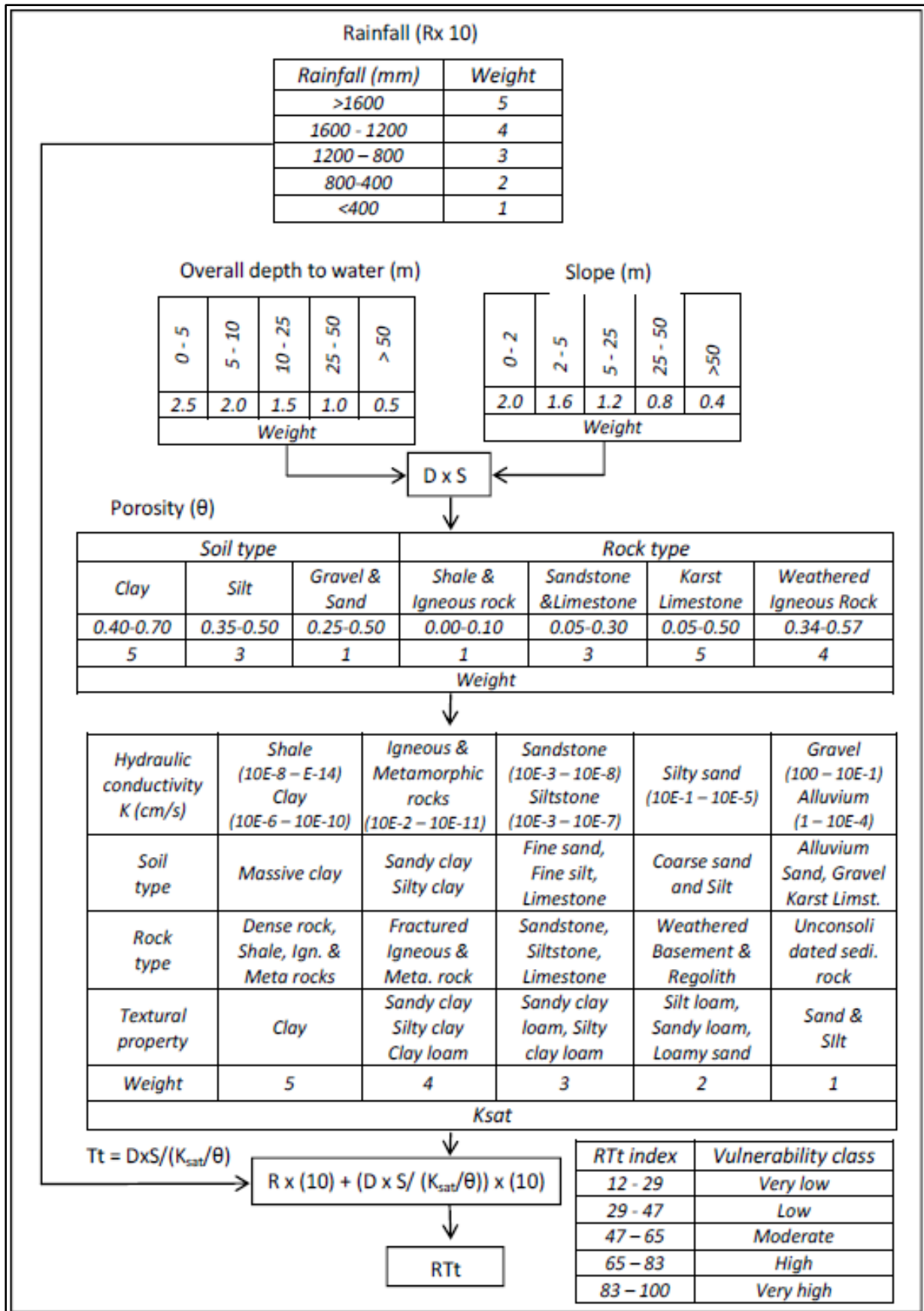


Figure 2.8 Objective and subjective criteria used in the rainfall travel time map (Oke et al., 2016)

2.3.10 Application of the Aquifer Vulnerability Index Method

The Aquifer Vulnerability Index (AVI) method is based on the concept of travel time of water and the movement of contaminants from the ground surface to the groundwater or aquifer. Originally developed by Van Stempvoort et al. (1993), the AVI method was designed to address some of the shortcomings of the DRASTIC method.

Experts often utilize the AVI method for small basin areas as it offers a simpler approach with fewer indices. For example, in the Densu River basin in Ghana, the AVI method was employed to formulate a groundwater vulnerability assessment. Similarly, in Figuig, Morocco, the AVI method was utilized to delineate vulnerability zones ranging from extremely low to extremely high (Putranto and Yusrizal, 2018b).

The AVI method evaluates the ease with which contaminants can migrate from the surface to the groundwater based on two primary factors:

- Thickness of the protective layers (typically soil and unsaturated zones) above the aquifer.
- Hydraulic conductivity of these layers, which affects how easily water (and potential contaminants) can flow through the subsurface materials.

The AVI is calculated by dividing the thickness of these protective layers by the average hydraulic conductivity, providing an estimate of how susceptible the aquifer is to surface pollutants. A higher AVI value indicates greater protection (less vulnerability), while a lower value suggests higher vulnerability to contamination (Van Stempvoort et al. 1993)

2.4 Limitations to groundwater vulnerability assessment methods

Groundwater vulnerability assessment methods vary in their applications, strengths, and limitations. The application of the principles of assessing intrinsic vulnerability can also be applied to assessing specific vulnerability assessments with the alteration of some of the parameters depending on the type of contamination for which the specific vulnerability is to be made (Chakraborty and Kumar, 2016; Ribeiro et al., 2017). As indicated in Table 2.1, there is no groundwater vulnerability assessment method that has the capacity to accommodate all the physical and hydrological characteristics. Likewise, no vulnerability assessment method is error proof.

While many process-based methods rely on simulations of the conditions and variables in the study area, statistical methods mainly rely on the determination of variables and their association with actual occurrence of pollutants in the groundwater. However, some of the limitations to statistical methods include the subjective selection of variables together with the insufficient water quality observations (Khosravi et al., 2018).

The parametric methods typically involve the analysis or the amalgamation of factors that control the movement or transport of contaminants from the ground surface into the saturated area. This results in having different vulnerability classifications across different locations. The main advantage of the parametric methods is that they can be applied over large areas when using factors such as rainfall and depth to groundwater, which makes them suitable for regional scale assessments (Thapinta and Hudak, 2003). However, like most groundwater vulnerability assessments, the major drawback for the parametric methods is their subjectivity in assigning numerical values to the descriptive entities and relative weights for the different attributes (Ducci and Sellerino, 2022; Li et al., 2022; Niedbalska, 2022).

2.5 Validation of groundwater vulnerability assessment methods

The validation of groundwater vulnerability assessments is used to affirm the suitability and efficiency of the groundwater vulnerability assessment method. Despite earlier arguments by Leonard et al. (1992), that groundwater vulnerability models are just epitomes of scientific hypotheses as they cannot be proven or validated, the use of validation techniques for groundwater vulnerability assessment has continued to grow. However, many researchers continue to use different techniques to validate proposed vulnerability methods as there is no universally agreed upon method for validating groundwater vulnerability.

Validating groundwater models is important for rooting out errors and improving weaknesses in groundwater vulnerability techniques. The most common methods of validation, as stated by Daly et al. (2002), Neukum et al. (2008), and Oke (2015), include bacteriological analysis, chemographs, hydrographs, tracer techniques, analogue techniques and calibrated numerical simulation techniques.

Establishing groundwater vulnerability due to NO_3^- is the most used validation technique, as NO_3^- is produced as a byproduct in many natural and anthropogenic

systems (Wei et al., 2021). Moreover, NO_3^- validation is considerably affordable and easy to operate compared to other validation techniques such as validations using chloride, dissolved oxygen, and borehole pump testing (Oke et al., 2016). The quality of data used on the groundwater vulnerability assessment method plays an important role in determining the outcome of groundwater vulnerability.

2.6 Validation using nitrate concentration

NO_3^- concentration can be used to validate groundwater vulnerability in many contexts. Jhariya et al. (2016) used NO_3^- concentration to validate the use of the DRASTIC model in the evaluation of aquifer vulnerability in the Tandula watershed, Chhattisgarh, India. In his validation, the NO_3^- concentration was found to be 82.35%, which was consistent with the findings of the DRASTIC model that has detected vulnerabilities of high as 23.72% and very high as 17.34%.

Ribeiro et al. (2017) used NO_3^- validation to assess the effectiveness of the incorporation of the land use index in the modified DRASTIC method against the original DRASTIC method on assessing the vulnerability assessment on diffuse agricultural pollution. The results obtained, and subsequent validation using NO_3^- concentration measured in situ, indicated the advantage of the susceptibility index over the original DRASTIC method, as the DRASTIC method revealed an underestimation of the vulnerability for almost the entire aquifer area.

Huan et al. (2012) used NO_3^- in their validation on the assessment of groundwater vulnerability to NO_3^- based on the modified DRASTIC model in the Jilin City of north-east China. Bonton et al. (2012) were able to validate their Water flow and Nitrate transport Global Model (WNGM) using NO_3^- concentrations in southern Quebec, Canada. Bonton et al. (2012) developed the WNGM method to simulate the impact of agricultural activities on NO_3^- concentrations in both an aquifer and municipal well. The WNGM method used the hydro-geosphere model to simulate a three-dimensional variably saturated groundwater flow and NO_3^- transport in the aquifer by using water and NO_3^- fluxes. The WNGM model was calibrated by reproducing water levels measured over a two-year period using a network of piezometers and NO_3^- concentrations measured in the municipal well across a ten-year period. The NO_3^- concentrations measured in the network of piezometers, however, showed greater variability than in the municipal well and could not be reproduced by the calibrated model.

The advantages of using NO_3^- for the validation of groundwater vulnerability assessments is that it is easily available and can be applied in many investigations, including mining, agriculture, and leachate contamination in waste sites. However, the natural presence of NO_3^- in many natural systems may compromise the reliability of NO_3^- (Mazdarani et al., 2023; Patel et al., 2022).

2.7 Validation using pump testing

Like NO_3^- , pump tests can also be used for validating groundwater vulnerability outcomes from different methods. The validation through pump testing utilises characteristics such as water quality, yield, and efficiency and is mostly applicable to boreholes or wells (Balasubramanian, 2017).

The use of pump testing, however, relies on the following assumptions:

- The geological formations must be horizontal and of limitless horizontal extent.
- The potentiometric surface of the aquifer must be horizontal prior to the start of the pumping.
- The potentiometric surface of the aquifer must not be changing with time prior to the start of the pumping.
- Groundwater flow must be in a horizontal direction.
- Groundwater should have a certain density and viscosity.
- The aquifer should be isotropic and homogeneous.

The advantages of applying and using the pump test for validation of groundwater assessments is that the analysis of groundwater levels and pumping rates measured during pumping tests provide some indication on the state and behaviour of an aquifer or groundwater system. These tests are also useful in providing valuable information that helps to understand the groundwater system, as the method can be applied to both confined and unconfined aquifers. However, the decisions should be based on a wider understanding of the regional geology, hydrogeology, and environment.

2.8 Summary

The literature review on the application of specific vulnerability to point source pollution has highlighted that there is no one-size-fits-all solution when it comes to groundwater vulnerability assessments. The context and nature of the contamination, as well as the

availability of resources, dictate the approach and methodology that is suitable for any groundwater vulnerability assessment. Parametric methods, which can also be classified as overlay or index methods such as DRASTIC, SINTACS, and GOD, have been widely used in assessing groundwater vulnerability to pollution as they are easy to modify and apply in identifying groundwater to contamination from specific sources. However, the subjectivity of the criteria utilised in selecting some of the parameters in parametric methods compromises their efficiency. Parametric methods were mainly developed to counter the limitations found in most process-based models and because of a lack of monitoring data required for statistical methods.

Compared to parametric methods, the use of numerical and statistical methods is limited. This is mainly because of the complexity that is involved in the calculations as the methods mainly rely on finding the probability of contamination of groundwater by pollutants. Despite having relatively high accuracy rates, compared to other groundwater vulnerability assessment techniques, numerical and statistical methods are usually utilised as surrogates for validating observations and outcomes from other groundwater vulnerability assessments. However, like many groundwater vulnerability assessment methods, statistical methods can only be applied to the geographic areas in which the data were collected and to regions where similar factors are associated with the likelihood of groundwater contamination.

This literature review has shown that although we can be able to identify the factors that cause groundwater contamination and design vulnerability models that incorporate these parameters, our ability to apply these models in real life contexts is considerably limited. Because of this, groundwater quality data should be used with some measurable degree of caution. As various methods for evaluating groundwater vulnerability to contamination are available, each has its own strengths and limitations. The validation of these methods is thus crucial in producing a more useful and accurate assessment.

2.9 Research gap

Various methods have been designed to evaluate the groundwater vulnerability to contamination; however, only a few of these methods can be employed to determine groundwater vulnerability from point sources. Since the development of the groundwater vulnerability concept by Aller and Margat in 1987, very few methods have focused on specific groundwater vulnerability assessment methods due to their complexities and

limited application. This study therefore sought to develop a reliable groundwater vulnerability assessment method that could examine the vulnerability of groundwater to contamination from a mine dump. The new groundwater vulnerability assessment method considers a set of parameters that have not only been applied in groundwater vulnerability assessments but also to other point source pollution sources. Unlike modified or adapted groundwater vulnerability assessment methods, the developed method was specifically designed to address point source pollution contamination, a gap that so many studies have not yet exploited. The newly designed groundwater vulnerability assessment method was tested in the Welkom area through the assessment of the groundwater vulnerability of the boreholes at the CUT campus to contamination from mine dumps.

Chapter 3

Physiography and Geology of the Welkom Area

3.1 Introduction

This study was carried out in the Welkom area, a town within the Lejweleputswa DM in the Free State Province of South Africa. For the purpose of this study, the physical, geographical, and geological features of the Welkom area under the Lejweleputswa DM are discussed in detail. These included descriptions of the climate, weather, rainfall patterns, recharge, and the nature of the geology, which include the rock and soil types found in the study area. The knowledge and background on the physiography and geology of the study area is critical in the development of a groundwater vulnerability assessment method as it lays out inputs on the surface water and groundwater conditions.

3.2 Description of the study area

3.2.1 Location, population, and economic significance of Welkom

Welkom is a mining city which is located in the north-eastern part of the Free State Province. According to Population Hub (2023), the estimated population of Welkom in 2023 was 667 288. Welkom is one of 50 cities in South Africa and ranks 16th on the overall population in cities. The city is located 250 km south of Johannesburg and 150 km north-east of Bloemfontein, which is the capital city of the Free State Province (John, 2012). Figure 3.1 shows the location of the Welkom area in relation to South Africa and the Free State Province.

Welkom, situated in the northwestern region of the Free State Province, serves as the capital city of the Lejweleputswa District Municipality. It shares its borders with the North West Province to the west and the Northern Cape Province to the north. The Lejweleputswa DM is one of the most mined districts in the Free State Province and one of the most mined regions in South Africa. Some of the big mining corporations working in Lejweleputswa DM are Harmony Gold, Petra Diamonds, and Gold Fields Limited (South African Venues, 2011). The mining sector accounts for approximately 31.5% of the district's gross domestic product and contributes roughly 23.8% to the total gross

domestic product of the Free State, positioning it as the second-largest contributor among the five district municipalities within the municipal district.

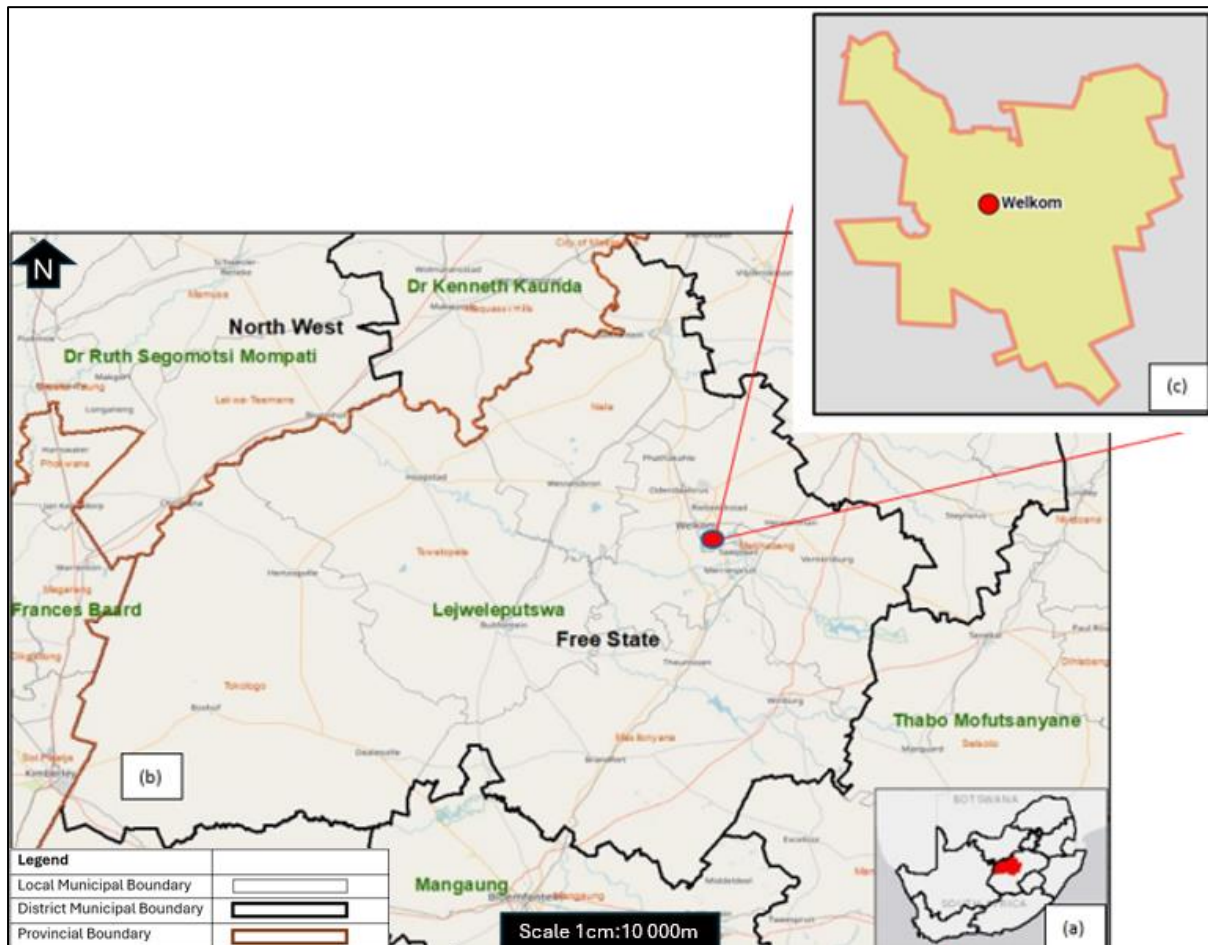


Figure 3.1 Location of the study area in South Africa and the Free State Province (a) Map of the Free State Province within South Africa; (b) Map of the Lejweleputswa District Municipality within the Free State Province; (c) Map of the Welkom area within the Lejweleputswa District Municipality (Google Maps, 2023)

The Welkom area stands out as one of South Africa's foremost gold mining regions, housing some of the largest goldfields in the country. However, the history of gold mining has over time resulted in many gold dumps, some of which are very close to human settlements and farmlands. Figure 3.2 shows a map of the study area with mine dumps across the Welkom area. This has affected the livelihood of the residents as some of their surface water and underground water sources have been contaminated and became undrinkable.

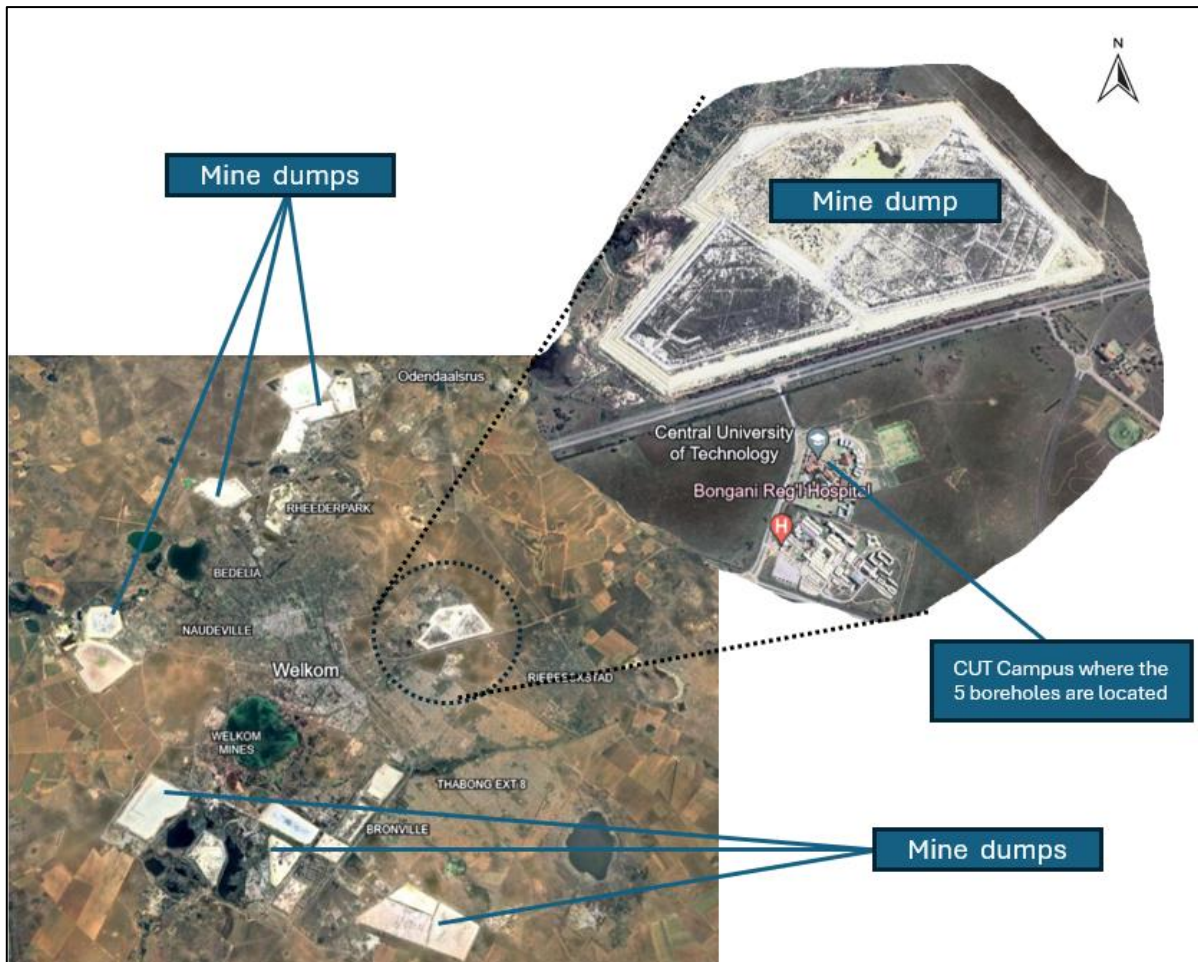


Figure 3.2 Map of the study area showing the mine dumps around Welkom and next to the CUT Welkom campus (Google Maps, 2023)

3.2.2 Climate and weather

According to the Köppen-Geiger classification system, South Africa belongs to a subtropical climate, which is characterised by average summer temperatures ranging from the mid-30 °C or higher (Peel, 2007). The average rain occurs mainly during summer and is in the range of 250–850 mm per annum, with more rain towards the coast. Extended periods of severe drought can occur far from the coast. In the northern parts frost is restricted to July, but can occur throughout the winter in the south and can be heavy around the mountains.

The Köppen-Geiger climate classification system (Figure 3.3) categorises the climate of the Welkom area as BSk (arid, steppe, and cold arid) (South African Weather Service [SAWS], 2023). This is because it is associated with the Highveld, and described as warm temperate, with hot summers and cold winters. During the summer months, the average

monthly temperature reaches approximately 32 °C, whereas in winter, it decreases to around 18 °C on average (Green Door Environmental, 2013).

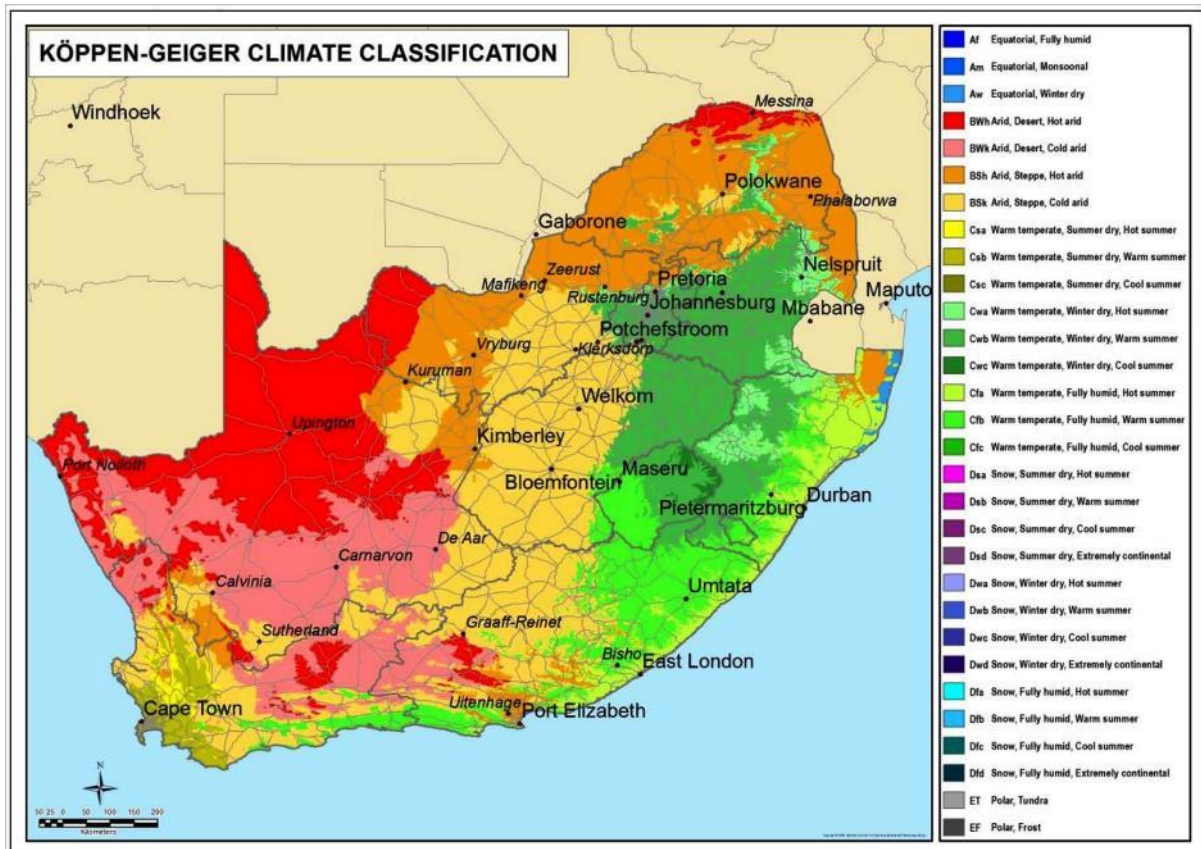


Figure 3.3 Köppen-Geiger climate classification of South Africa (Peel et al., 2007)

3.2.2.1 Precipitation

Precipitation in Welkom occurs mostly within the summer months, with an annual average of 490 mm (SAWS, 2023). Figure 3.4 shows the average chance of precipitation in the Welkom area across the year with an average of 37.5% of the rainfall in the summer season (December–March), 21% falling in the autumn season (April–May), 2% falling in winter season (June–August), and 21% in spring (September–November). Depending on other conditions such as soil, rock, and aquifer type, the precipitation pattern determines the extent of groundwater vulnerability to contamination from mine dumps in the summer months as precipitation has the effect of transporting and moving contaminants through the soil, and rocks to the groundwater. Similarly, when the precipitation is low in the winter months, it is expected that the groundwater vulnerability to contamination will be very low.

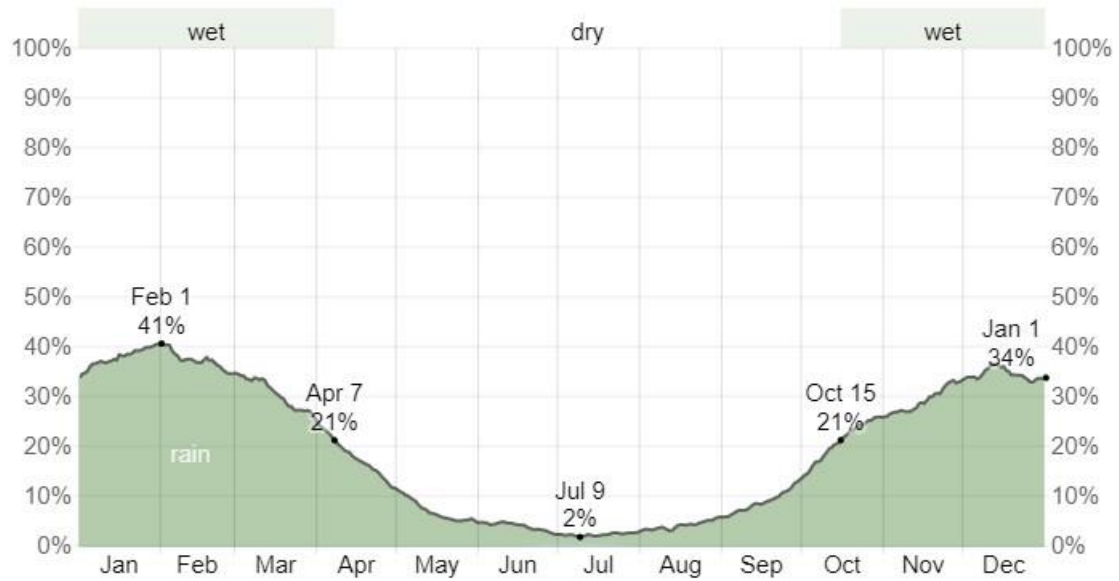


Figure 3.4 Average monthly chance of precipitation in the wet and dry season in Welkom (Climate Data, 2024)

When compared to the average precipitation patterns in the Free State Province, the average precipitation in the Welkom area was lower by margins close to 3% (SAWS, 2023). This means that depending on other conditions, the vulnerability of the Welkom area to contamination may be slightly low, compared to other areas within the Province. Figure 3.5 shows the average rainfall patterns of the Free State Province from 1921 to 2022.

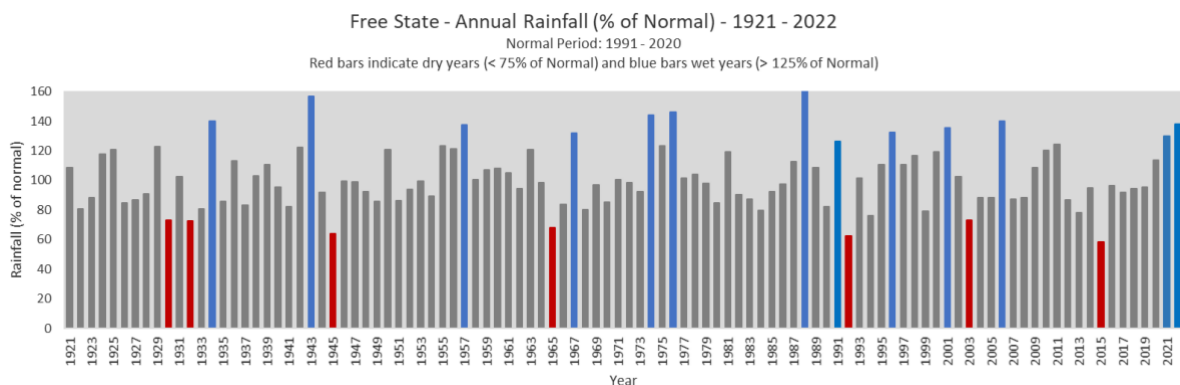


Figure 3.5 Average annual rainfall distribution for the Free State Province, 1921–2022 (SAWS, 2023)

3.2.2.2 Recharge

According to Musekiwa and Majola (2013), the average annual recharge rate for the Welkom area lies between 5 mℓ and 10 mℓ. This is mainly due to the low average rainfall received in the area, together with other conditions that affect recharge such as

vegetation, soil, and rock structures. Figure 3.6 illustrates the recharge map of South Africa, including the Welkom area. Compared to other regions in South Africa, the recharge in the Welkom area is next to the minimum. As discussed, this can be due to many factors, including rainfall patterns, soil type, vegetation, and elevation. Depending on the soil and other surface conditions, studies have shown that light to moderate rainfall patterns usually result in less surface run-off and more infiltration and recharge, while heavy rain results in more surface run-off, less infiltration, and less recharge. All conditions will be considered in the assessment of groundwater vulnerability in the study area.

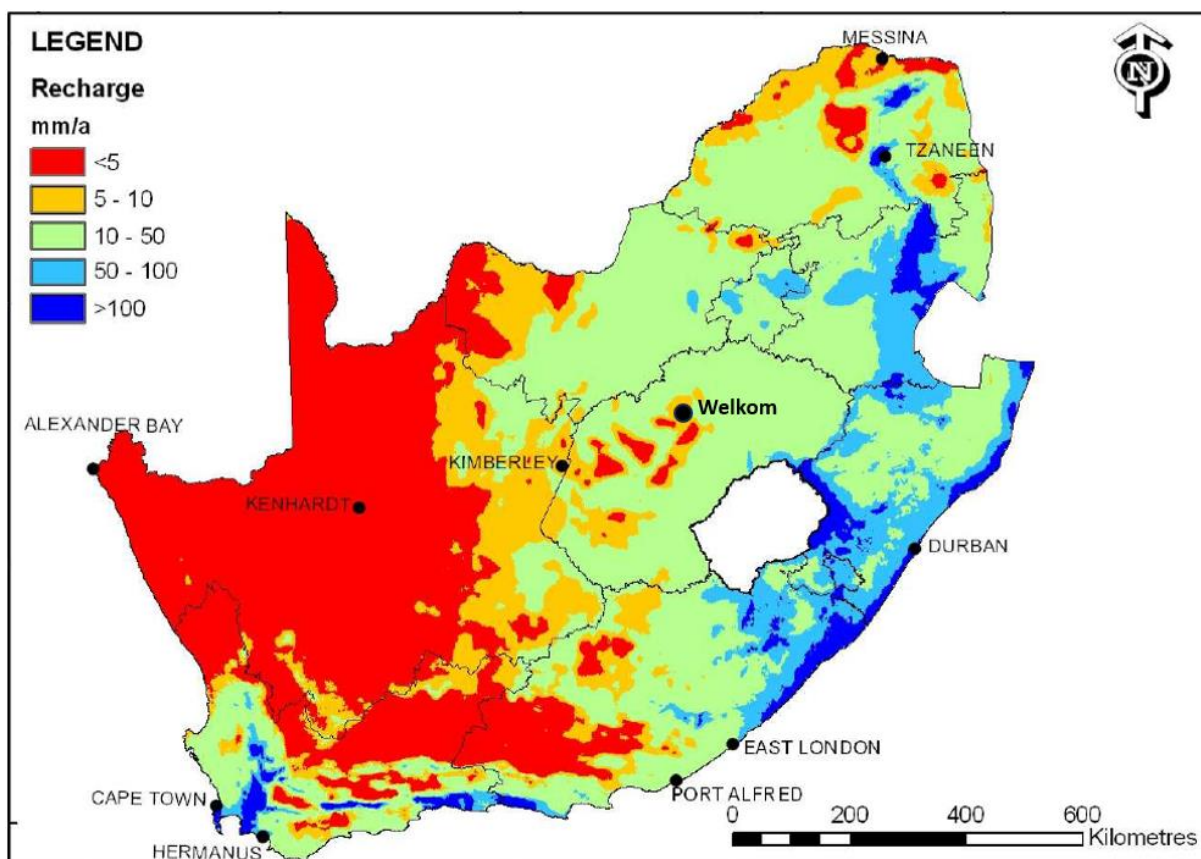


Figure 3.6 Recharge map of South Africa (Musekiwa and Majola, 2013:156)

3.2.2.3 Temperature

The average daily temperatures are responsible for evaporation. In hot and arid climates, the daily temperatures may affect the saturation of the topsoil, which may affect infiltration and surface run-off. When temperatures drop to levels below zero in winter, some areas in Welkom become frozen and this may affect surface water movement, especially in winter. The average temperatures in the Welkom area drop between the

months of April and September, while the temperatures get warmer between January and March, as well as between September and December. Figure 3.7 shows the hourly temperature ranges within the Welkom area.

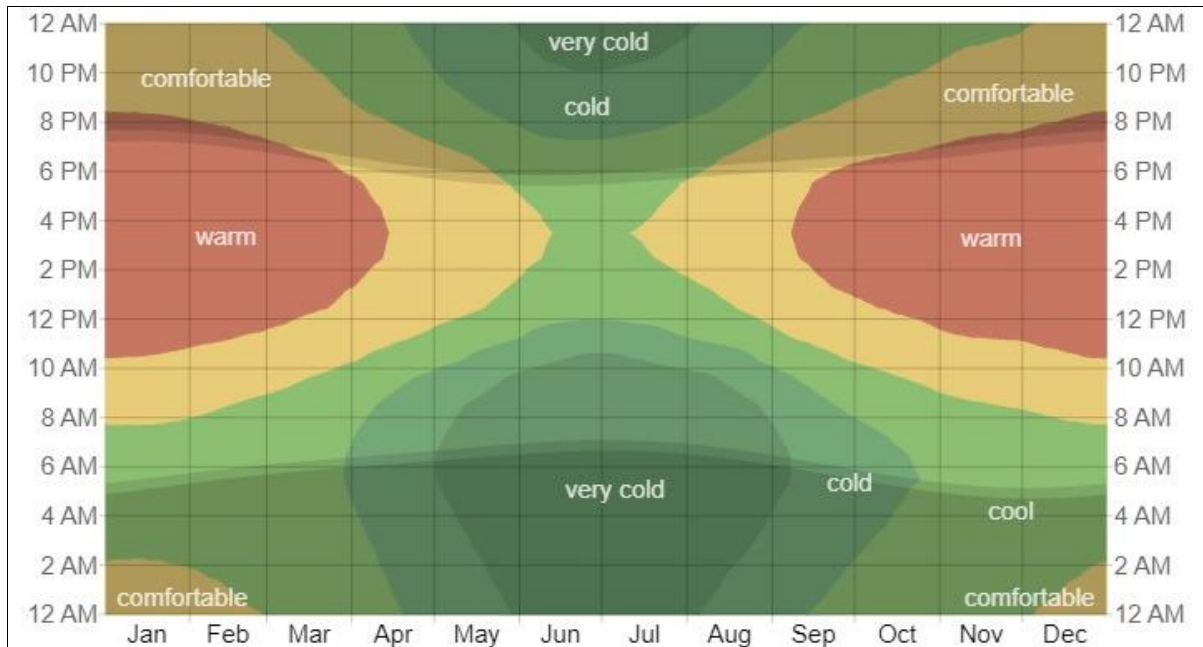


Figure 3.7 The Welkom monthly average hourly temperature, colour-coded into bands (Climate Data, 2024)

3.2.3 Classification of the geology in the Welkom area

The geology of the Free State Province can be broadly categorized into the Witwatersrand Supergroup, the Karoo Supergroup, and the Ventersdorp Supergroup. The majority of regions within the Free State Province are situated within the Karoo Supergroup, which is subdivided into the Beaufort and Ecca Groups. Sedimentary rocks, such as sandstone, shale, and mudstone, predominate in these groups. The Beaufort Group is primarily characterized by sedimentary deposits spanning from the Middle Permian to the Middle Triassic Period (Department of Environmental Affairs, 2010).

The study area's geology is also encompassed within the Witwatersrand Supergroup. Figure 3.8 shows the map of the Welkom Goldfields as part of the Witwatersrand Basin.

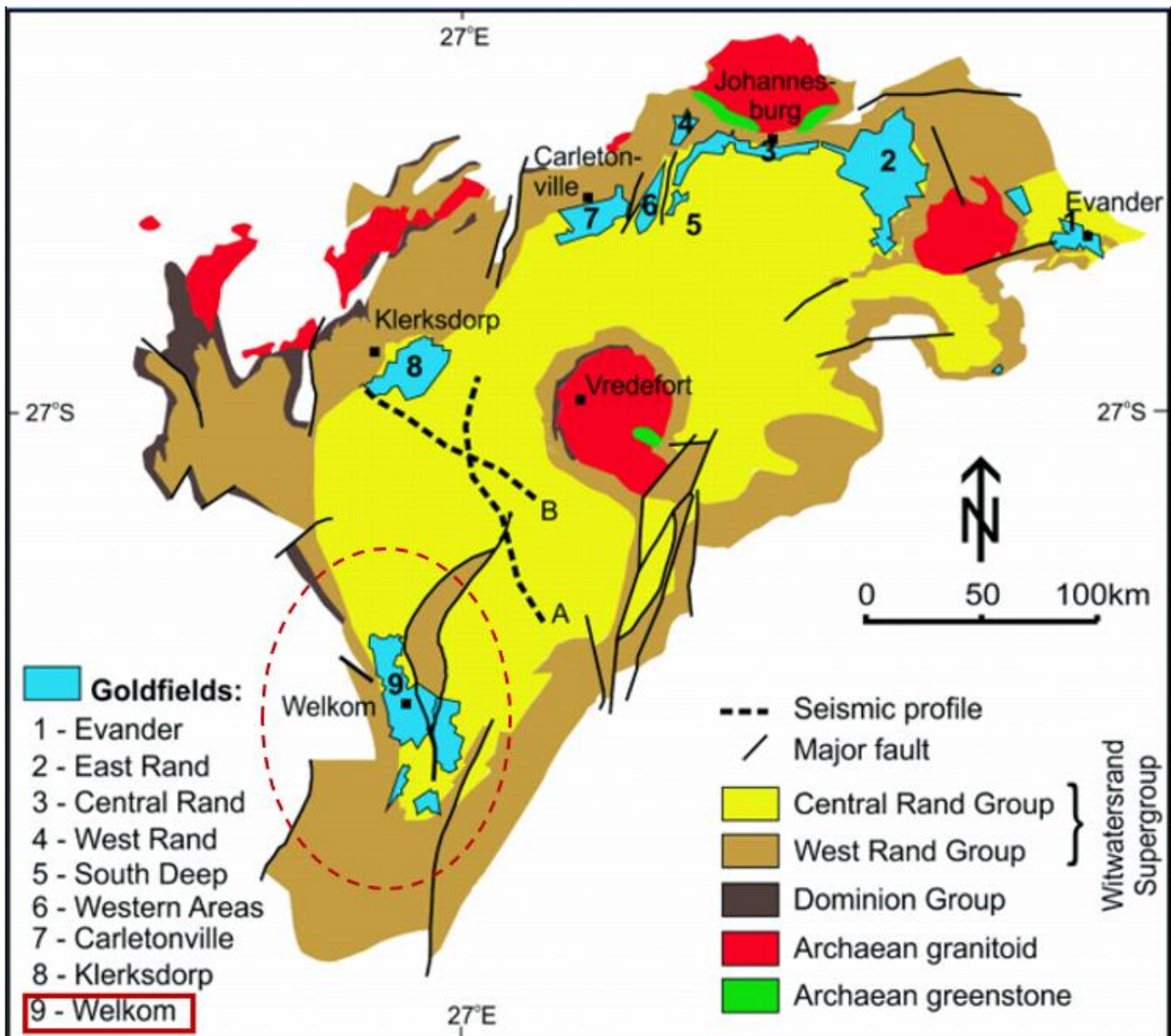


Figure 3.8 Welkom Goldfields are classified as part of the Witwatersrand Supergroup (Frimmel et al., 2005; Musekiwa and Majola, 2011)

The Witwatersrand group can be subdivided into the Central Rand Group (CRG) and the West Rand group, both forming layers of the bedrock in the study area (Musekiwa and Majola, 2011). The formation of the Witwatersrand Basin was a result of successive crustal plate movements from the north and west within a foreland basin setting. Deposition initially occurred in a shallow marine environment, characterized by the accumulation of shale and quartzite, with notable iron-rich shale horizons forming the West Rand Group. The subsequent deposition of the predominantly sandy CRG occurred as the basin regressed and gradually filled. The gold mineralization within the CRG primarily consisted of detrital gold,

predominantly composed of sulphides and uraninite. Initially, these deposits were concentrated alongside placer minerals like chromite and zircon (Eriksson et al., 2006).

3.2.4 Stratigraphy of the Witwatersrand Basin

According to the South African Committee for Stratigraphy (1980), the Witwatersrand Supergroup is composed of the West Rand Group (WRG) and an upper CRG. The CRG serves as a standard reference section, allowing for correlation of strata across all goldfields. However, in the Welkom area, local nomenclature is primarily employed, sometimes merging formations where clear distinctions cannot be made.

3.2.4.1 Central Rand Group

The CRG contains most of the mineralisation in the Witwatersrand Group. The sequence is predominantly composed of sandstone formations, including quartzites, quartz-pebble conglomerates, quartzwackes, and minor shale layers (Beukes and Nelson, 1995). The CRG is subdivided into the lower Johannesburg subgroup and the upper Turffontein subgroup. In the Welkom Goldfields, these subgroups are separated by the Booyens Shale formation (Maynard and Klein, 1995). The CRG consists of multiple formations, which, despite differences in thickness, can generally be traced and correlated, with a few exceptions. The gold-bearing conglomerate reefs are often found in clusters known informally as reef groups. These reef groups are situated on prominent unconformity surfaces that can be traced throughout the entire basin (Dankert and Hein, 2010).

Figure 3.9 shows the different formations of the CRG, including the one for the Welkom area.

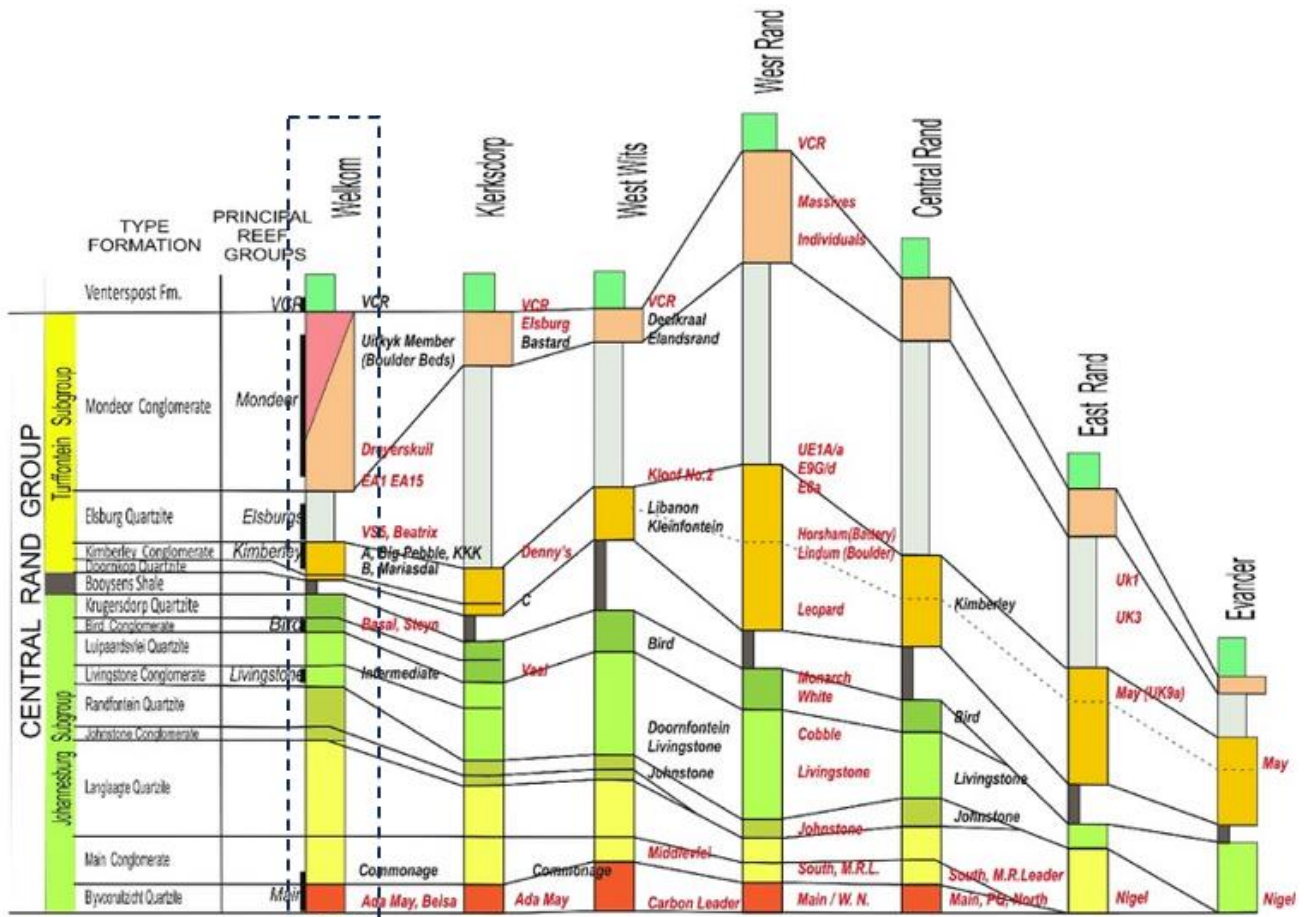


Figure 3.9 The stratigraphy of the Central Rand Group illustrates the subgroups, formations, and gold-bearing conglomerate reefs across the seven major goldfields (McCarthy, 2006; Pretorius, 1974; South African Committee for Stratigraphy, 1980)

3.2.4.2 West Rand Group

The sediments of the WRG were deposited due to rapid episodes of mechanical subsidence, which occurred in a pull-apart basin formed under trans-tensional rifting conditions (Maynard and Klein, 1995). Roughly equal proportions of quartzites and iron-rich shales dominate the area (Pretorius, 1974; Gutzmer and Beukes, 2020). A significant number of quartzites represent sand bodies from shallow marine shelves (Beukes and Nelson, 1995; Eriksson et al., 2006; Bumby and Eriksson, 2021). The WRG includes the lower Hospital Hill Subgroup, the Middle-Government Subgroup, and the upper Jeppestown Subgroup. The shales within the WRG are distinguished by the occurrence of interlayers containing magnetite. These layers served as important markers during deep basin exploration due to their magnetic properties. Figure 3.10 indicates the

broader geological map of South Africa, showing that the Welkom area is largely dominated by shale and sandstone rocks.

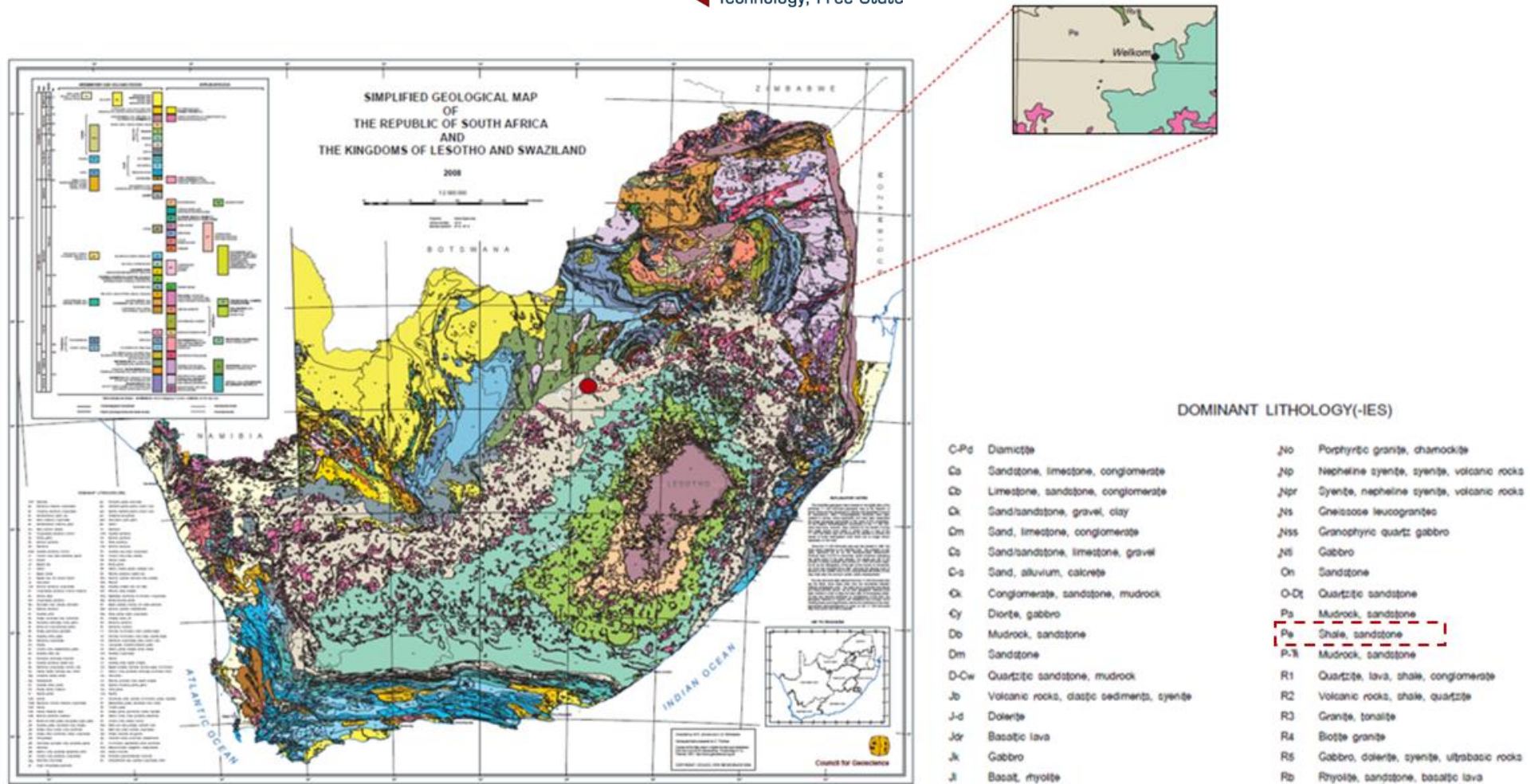


Figure 3.10 Geological map of South Africa showing dominant lithologies(Johnson and Wolmarans, 2008)

Despite extensive mining and geological research conducted over the years, the intricate structural complexity of the Witwatersrand Basin has not been comprehensively compiled to date.

The primary source of structural information has been derived from the mapping and consolidation of data collected from underground operations by mine geologists, surveyors, and samplers. Detailed structural maps of the major goldfields have been created, revealing complex faulting, and folding, particularly along the northwestern margin of the basin (Dankert and Hein, 2010).

3.2.5 Characteristics of shale and sandstone rocks

In Welkom, South Africa, the predominant type of shale is often associated with the Karoo Supergroup, particularly the Ecca group (Department of Environmental Affairs, 2010). The three most common types of shale in Welkom are the Carbonaceous shale, the Siltstone shale, and the Mudstone shale. Mainly distributed between the Palaeozoic to the Cenozoic ages, shale rocks have a wide range of colour spectrums from white to grey to green and black, depending on their composition and environments (Boggs, 1995: 2009). The Carbonaceous shale is rich in organic material, typically dark grey to black in colour and contains carbon in the form of plant debris. The siltstone shale contains more silt-sized particles than typical shale and tends to have a slightly coarser texture and can be lighter in colour. Mudstone shale is a type of shale that has a very fine-grained texture, composed mostly of clay minerals with minimal organic content and tends to be grey or brownish in colour (Krumbein and Sloss, 1963; Tucker, 2001).

The sandstone in Welkom is primarily composed of sand-sized mineral grains, mostly quartz but may also include feldspar and other minerals. Sandstone has a medium to coarse-grained texture, with visible particles that are often cemented together by minerals like silica, calcium carbonate, or iron oxide. Sandstone can range in colour from white, yellow, and brown to red, depending on the composition of the cementing material (Okeke and Okogbue, 2011). Both shale and sandstone play crucial roles in the geology of the Welkom area and hence in the vulnerability of groundwater to contamination.

Figure 3.11 shows the shale stone and sandstone layered formations characteristic of the Welkom area.



Figure 3.11 Images showing (a) shale stone, (b) layered shale stone formation, (c) sandstone and (d) layered sandstone formations (Author, 2024)

Unlike shales, sandstones are sedimentary rocks that are mainly composed of sand-sized mineral particles or rock fragments. The colour of sandstones varies considerably depending on their mineral components. Sandstone colours range from brown, red, purple, pink, green, to light grey (Reading, 1996; Nichols, 2009). Sandstones are usually formed around areas where there is a lot of accumulation of sand such as in riverbeds, deserts, or coastal areas. Like shale, the mineral composition of sandstone has different mineral components, which include clay minerals, quartz, mica, and feldspar. These minerals act as cement by holding the sand grains together with the degree of rock strength dependent on the composition of its constituent minerals (Tucker, 1996; 2001). In most cases, sandstone is very porous, thus allowing fluids such as water and oil to pass through. In regions where sandstone predominates, the infiltration rate is high, rendering the groundwater highly susceptible to contamination. The environment where the sandstone is located is important in determining its characteristics, which includes its grain size, sorting, and composition, and also its rock geometry (Schlumberger, 2023).

3.2.5.1 Porosity of shale and sandstone

The porosity of rocks has a huge effect on the underlying groundwater vulnerability. Depending on the environment, the porosity of both the shale and sandstone varies and influences the groundwater vulnerability to a greater extent. The groundwater vulnerability assessment method; developed in this study, depends on the characteristics of the rock formation in the Welkom area. Figure 3.12 shows the granular properties of the shale and sandstones and how they are affected by burial and compaction.

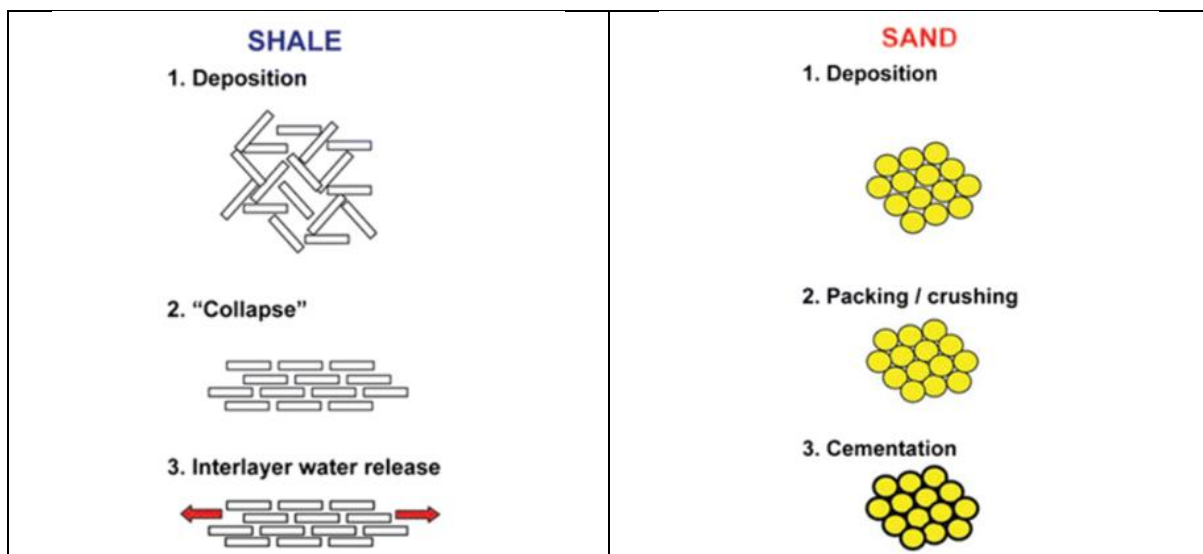


Figure 3.12 Three stages of compaction affecting the porosity of shale and sandstone (Avseth et al., 2003)

The rate of shale porosity decreases with an increase in burial depth or compaction. This may be due to the decreasing shale permeability and increasing water viscosity. For a given increase in overburden pressure, the pressure applied to a given grain-to-grain contact area decreases with increasing burial distance as the contact area increases with compaction and grain deformation (Reading, 1996; Nicols, 2009). These observations indicate that the shale compaction is primarily controlled by physical forces acting on sedimentary rocks. On the contrary, the rate of sandstone porosity reduction is known to be relatively constant throughout the burial diagenetic history in many regions. For a given increase in overburden pressure, the grain-to-grain contact area decreases yet the rate of porosity reduction for the sandstones remains relatively constant whether they are at a shallow or deep burial depth. This suggests that, for the sandstones, the physical forces are of secondary importance in reducing the porosity

(Avseth et al., 2003). Figure 3.13 shows the change in porosity of shale and sandstone as a function of compaction.

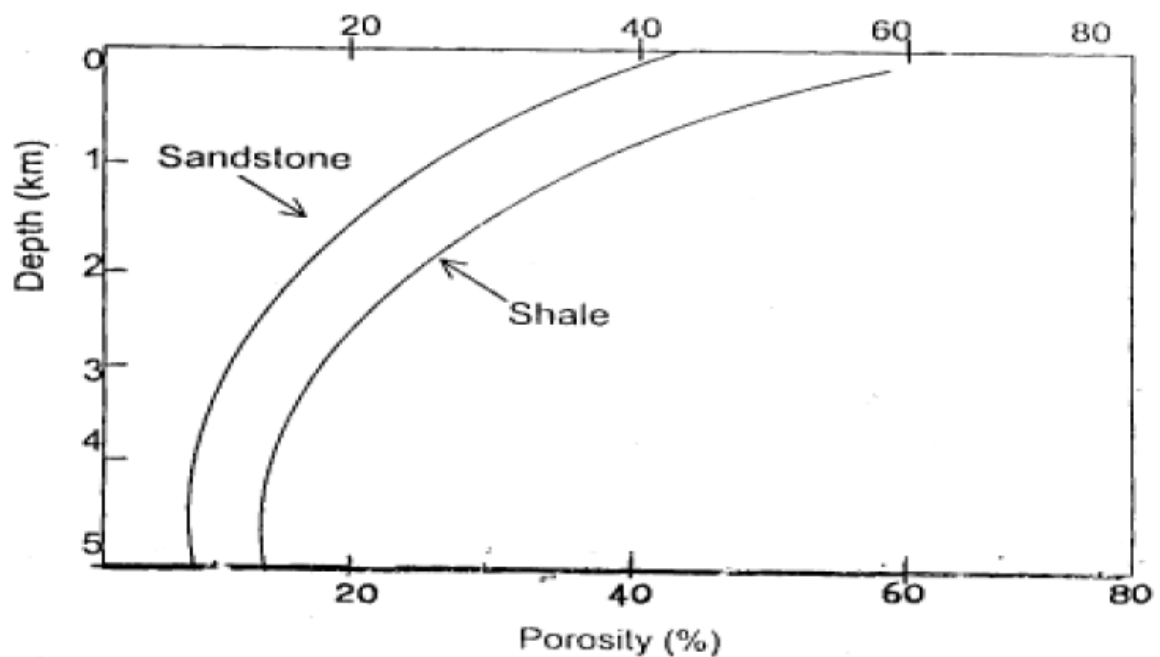


Figure 3.13 Curves indicating the change in porosity for shale and sandstone as functions of burial depths or compaction (Boggs, 1995)

The waste rocks composing the shale and sandstone in the mine dump under investigation is shown in Figure 3.14. It is suspected that the active weathering and leaching of minerals is ongoing.



Figure 3.14 Waste rocks (mine dumps) seen from sports field at CUT Welkom campus (Author, 2024)

3.2.5.2 Classification of the soil type in Welkom

According to Masindi and Muedi (2018), soils are the major sinks by which contaminants enter the environment. Soil type plays an important role in influencing groundwater vulnerability as it either enables or disables the infiltration and movement of water between the surface water and groundwater. The texture and chemical composition of soil determines its ability to protect the underlying groundwater aquifers.

The contaminants released into the soil may accumulate and have lasting detrimental effects on the groundwater resources. At the CUT campus in Welkom, the groundwater although available, has never been used for drinking water because of its aesthetic properties and suspicions of contamination from nearby mine dumps. Studies conducted on the topsoil in a gold mine dump in Western Australia revealed abnormally high levels of As and other heavy metals such as Pb, Cd, Cobalt (Co), Cr, and Ni (Abraham et al., 2018). Liang et al. (2017) found abnormally high concentrations of Pb, Zn, Cu, Cd, Ni, and As in the top soils near a mining dump in Guangdong, China. In South Africa, Olobatoke and Mathuthu (2016) found abnormally high concentrations of As in soils near a gold mine dump in Johannesburg, with As concentrations as high as 235 mg/kg, far exceeding the recommended industrial minimum concentration of 150 mg/kg required by the South Africa Soil Quality Standards.

In South Africa, soil quality standards aimed at remediation of contaminated soils is governed by legislation. According to legislation, the soil screening values (SSVs) must be used as guidelines for screening purposes against heavy metals in areas where soil needs to be remediated from contamination activities such as mining.

Tables 3.1 and 3.2 shows the SSVs used for metals and anions.

Table 3.1 Soil screening values for metals

Parameter	Units	SSV1 All land-uses protective of the water resource	SSV2 Informal residential	SSV2 Standard residential	SSV2 Commercial residential	Protection of ecosystem health
Metals and metalloids						
Arsenic	mg/kg	5.8	23	47	150	580
Cadmium	mg/kg	7.5	15	32	260	37
Chromium (III)	mg/kg	46 000	46 000	96 000	790 000	n/a
Chromium (IV)	mg/kg	6.5	6.5	13	40	260
Cobalt	mg/kg	300	300	630	5 000	22 000
Copper	mg/kg	16	1,100	2 300	19 000	16
Lead	mg/kg	20	110	230	1,900	100
Manganese	mg/kg	740	740	1 500	12 000	36 000
Mercury	mg/kg	1.0	1.0	1.0	6.5	4.1
Nickel	mg/kg	91	620	1 200	10,000	1,400
Vanadium	mg/kg	150	150	320	2,600	-
Zinc	mg/kg	240	9,200	19,000	150,000	240

Source: Department of Environmental Affairs (2010)

Table 3.2 Soil screening values for anions

Anions	Soil screening level (mg/kg)
Chlorides	12 000
Fluorides	30
Nitrates–nitrites	120
Sulphates	4 000

Source: Department of Environmental Affairs (2010)

3.2.5.3 Assessment of soil quality in the study area

The predominant natural soil covering much of the Welkom area consists of duplex soil, exhibiting contrasting textures between soil horizons. Soil colour varies between red, yellow, brown, and grey hues (Green Door Environmental, 2013). However, the soil found in areas surrounding the mine dumps ranges from dark red to a sandy whitish colour. Soil characterisation studies conducted by Belle (2021) and Musekiwa and Majola (2013) confirmed from ground observations that the areas surrounding the mine dumps consisted of white sandy loam soils. Figure 3.15 shows images of the whitish and reddish top soil on top of and around the mine dump.



Figure 3.15 Image showing the white–reddish sandy soil covering the surrounding mine dump
(Author, 2024)

The soil quality in the study area was determined through reviews on previous studies that were conducted in the Welkom mining and former mining areas with mine dumps. A Study conducted by Belle (2021) on 30 soil sampling sites (with 15 cultivated and 15 uncultivated sites) around the Matjhabeng Local Municipality, including Welkom, assessed the soil quality by using indicators. The soil quality indicators included physical parameters such as electrical conductivity (EC) and pH, together with heavy metals and metals such as As, Pb, Cd, Cr, Fe, Cu, and Zn. The findings from the study indicated that both the cultivated and uncultivated soil samples close to mine tailings were contaminated with high concentrations of Pb, Cd, Ni, Fe, Cr, Cu, and Zn. Soil samples taken from zones close to the mining area indicated high concentrations of contaminants, compared to zones situated far from the mine dumps. Additionally, soils close to the mine dump indicated very high EC values and very low acidity levels.

The analysis of soil samples further indicated high concentrations of metals such as Cu, Fe, Cr and Zn in uncultivated soil samples, particularly during the summer seasons. The nature of these contaminants are often linked with gold mine tailings (Bokar et al., 2020; Wu et al., 2014).

Calcium is one of the major elements found in natural environments (Hao and Jiang, 2015). It is predominantly found as a constituent to various minerals and soils, however,

apart from its natural occurrence, high Ca concentrations can be an indication of soil pollution and contamination from mining activities. The regular liming operations conducted by mines to neutralise the high soil acidity uses lime water consisting of calcium hydroxide and calcium sulphate. This releases more Ca into the soil. In his studies on assessing the concentrations of mining pollutants, Ebenebe (2016) sampled soils across six towns in the Lejweleputswa DM to test for calcium concentrations linked to mining activities. The six towns included Welkom, Allanridge, Brandfort, Odendaalsrus, Virginia, and Theunissen. The results indicated that the concentration of soils in Welkom were significantly higher than the concentrations of soils in the other five towns. Figure 3.16 shows a comparison of the calcium concentrations across different towns in the Lejweleputswa DM.

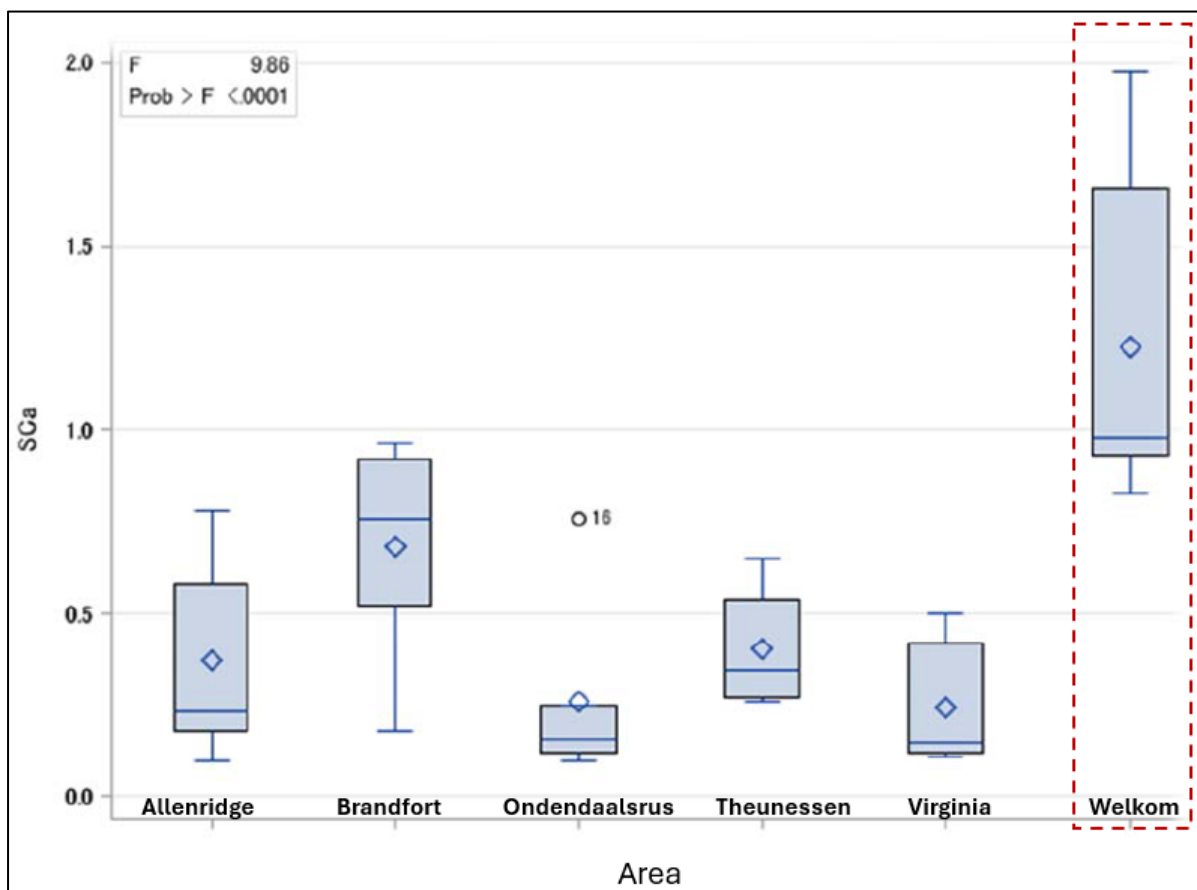


Figure 3.16 Percentage of calcium collected from six towns in the Lejweleputswa District Municipality (Ebenebe, 2016)

As shown in Figure 3.16, the soils in the Welkom area are characterised by a lot of Ca from mining activities. This also indicates the greater extent to which the soil is exposed to contaminants from the nearby mine dumps, and hence, the finding of a wide range of

heavy metals and low acidity levels. The low acidity in the Welkom soil may have resulted from the oxidation of pyrite in the mine tailings. This oxidation process releases acids which are transported from the mine dumps into the surrounding soils (McCarthy, 2011; Tutu et al., 2008). According to Wong et al. (2001), the low soil acidity may also affect the growth of microbes in the soil and surrounding water sources.

3.3 Summary

The physiography and geology of the Welkom area is unique to that of some former mine towns in the Free State and around South Africa. The relatively low annual precipitation patterns and high summer temperatures do not theoretically favour an increased risk of groundwater vulnerability in the study area, although some studies have shown that low and light precipitation can contribute to more groundwater recharge as opposed to heavy precipitation (Oke, 2015). The presence of the shale and sandstone rock types in the Welkom area may be a possible leading factor in the overall assessment of groundwater vulnerability as these rock types favour more infiltration than runoff. Additionally, the sand and loam soils characteristic of the Welkom area are also more prone to infiltration and less runoff; hence, contributing to the exposure or vulnerability of groundwater to contaminants. Additionally, the low levels of acidity together with the high levels of toxic heavy metals found in soils in the study area may have played a huge role in exposing the groundwater to contaminants through leaching of these contaminants from soil into the groundwater (Ebenebe et al., 2017). The information and data provided in this chapter will form the basis for the development of a new groundwater vulnerability assessment method.

Chapter 4

Hydrogeological Characterisation of the Study Area

4.1 Introduction

A site hydrogeological characterisation process was conducted to develop an understanding of the geologic and hydrogeologic properties of the study site. According to Benson and Yuhr (2016), site characterisation in groundwater vulnerability evaluations involves investigating the soil, rock, and groundwater properties to better understand the nature of the anthropogenic activities that impact the study site. Cook (2003) defined site characterisation as the activity needed to gather the hydrogeological information, and design effective remedies for contaminated soils and groundwater.

For this study, the soil and rock characterisation has been covered in Chapter 3 and this chapter will only focus on hydrological characterisation of the study site. The hydrological characterisation of the Welkom study site needs to be comprehensively and accurately determined in line with the objectives of the study. For this study, it is important for the characterisation process to provide as much qualitative and quantitative data and information that can be used in developing a vulnerability assessment method. The hydrological data were obtained through the following:

- Water quality characterisation.
- Detection of fracture points in aquifers.
- Determination of groundwater flow direction.

The characterisation of each of these elements is essential in understanding the extent to which the groundwater has been contaminated by the mine dump and hence in mapping its vulnerability. The water quality characterisation involved the calculation of the WQI of the groundwater using five boreholes located at the CUT Welkom campus to determine the extent of contamination of the groundwater, and hence to determine the groundwater vulnerability over a three-year period. It is through the water quality characterisation that the link between the contaminants from the mine dump can be traced to the borehole water at the study site.

Determining the aquifer type is useful in identifying groundwater flow movements in relation to contamination, while the soil quality determination is necessary for drawing out comparisons and inferences in water quality differences between the contamination site and the groundwater. The determination of the aquifer fracture points was conducted to assist in determining the certainty of the groundwater flow direction and to some extent the rate of groundwater recharge.

It should, however, be noted that water quality, detection of fracture points and groundwater flow direction are not exhaustive of all the parameters that can be used to assess groundwater vulnerability to contamination. Pump testing and borehole logging techniques can also be used for the further characterisation of groundwater to contamination from mine dumps. Relying on only three parameters to characterize the water may not be suitable for all mining environments. However, given that this study was conducted in a shallow unconfined aquifer, the hydrological parameters were deemed sufficient. Additionally, the study intentionally adopted a simplified approach, utilizing parameters based on easily accessible data to streamline the analysis.

4.2 Water quality characterisation: Assessing the drinking water quality indices of boreholes and surface water close to a mine dump in Welkom, South Africa

Access to safe drinking water is a basic human entitlement (WHO, 2022). The contamination of groundwater sources poses a worldwide challenge, impacting both human health and the environment (Satheeskumar et al., 2020). Particularly in regions near mining sites or tailings, the investigation of groundwater and surface water pollution is essential, given the potential health risks to humans and animals (Ding et al., 2020).

Various countries and international organizations have established statutory regulations to safeguard human health by setting standards for the quality of drinking water. In South Africa, the South African National Standard 241 (SANS 241) outlines drinking water quality guidelines for potable water. Additionally, the WHO (2022) offers water quality guidelines to protect the global population in regions where specific drinking water standards are not defined. Overall, these standards emphasize the physical, chemical, and microbiological parameters and concentrations essential for determining the suitability of drinking water in accordance with public health regulations.

The Water Quality Index (WQI) was devised to evaluate drinking water quality standards, encompassing various factors (Zhou et al., 2020). Originally introduced by Horton (1965) and further refined by Brown et al. (1970), the WQI underwent modifications in 1972 and evolved into different variations to suit diverse environments (Malan et al., 2003; Wertz and Shank, 2019). Most WQI methods utilize physical and chemical parameters to compute the index in diverse manners (Lumb et al., 2011). Employed to gauge drinking water quality in line with regulatory standards, the WQI serves as a metric that amalgamates the cumulative impact of numerous water quality determinants on overall water quality. Essentially, the WQI functions as a mathematical instrument to condense extensive water quality data into a single numerical value, furnishing water quality managers with the means to assess the quality and potential applications of a given body of water. The incorporation of various water quality parameters into the WQI hinges on the context, objectives, and methodology of the investigation (Water Research Commission, 2017).

The analysis of the groundwater quality is essential in tracing the extent of contamination of the borehole water from the mine dump contaminants. Analysing groundwater quality is crucial for tracking the extent of contamination from mine dump contaminants in borehole water. Unlike surface water, changes in groundwater quality occur gradually over time, rather than rapidly (Department of Water Affairs and Forestry (DWAF), 2000). Once groundwater is contaminated, the restoration of its quality is challenging and costly. According to Odonkor and Ado (2013), the quality of groundwater reflects on the inputs from the atmosphere, soil, and rock reactions as well as from point and non-point pollutant sources such as mining, farming, domestic and industrial wastes. This pollution is mainly attributed to nutrient, chemical and pathogen loadings into groundwater because of natural and anthropogenic point source and non-point source activities (Environmental Protection Agency, 2003).

This study aimed to examine the drinking water quality of groundwater sourced from five boreholes in Welkom, alongside an open water effluent stream near the mine dump, identified as a potential contaminant source. The assessment employed the SANS 241 (SANS, 2015) and WHO (2022) water quality guidelines to evaluate the quality of water from these sources.

4.2.1 Materials and methods

Groundwater from the boreholes at the CUT Welkom campus was periodically sampled and analysed during the summer and autumn seasons of 2021 and 2023 targeting the months of December and April respectively. To capture variations in water quality throughout the seasons, the grab method was employed to directly collect samples from six designated sampling sites, with each site sampled at least three times per season. Among these sites, five were boreholes labelled as B1 to B5, while the sixth site (S6) was an open water effluent stream situated at a distance of no less than 100 meters from a nearby mine dump. The boreholes' accessibility and proximity to the mine dump rendered them suitable sampling locations for the study. The sampling points' locations across the Welkom CUT campus are depicted in Figure 4.1, with corresponding coordinates provided in Table 4.1.



Figure 4.1 Sampling points for the groundwater and mine dump surface water (Adapted from Google Maps, 2023)

Table 4.1 presents the geographic coordinates and elevations of the sampling points

Table 4.1 Geographical coordinates of the sampling points

Sampling point*	Description	Longitude	Latitude	Elevation (m)
B1	Borehole 1	27° 56' 57,9" S	26° 47' 05,7" E	1 396
B2	Borehole 2	27° 56' 58,6" S	26° 47' 05,5" E	1 391

B3	Borehole 3	27° 56' 52,4" S	26° 47' 08,6" E	1 386
B4	Borehole 4	27° 56' 57,4" S	26° 46' 58,4" E	1 397
B5	Borehole 5	27° 56' 57,4" S	26° 46' 55,3" E	1 399
S6	Effluent stream from mine dump	27° 56' 52.7" S	26° 46' 41.1" E	1 403

* B1 to B5 are boreholes 1 to 5; S6 is Sampling point.

The sampling point on the mine dump was at a higher elevation than the others, which may have affected surface runoff, potentially leading to contamination of the boreholes.

The physical, chemical, and microbiological elements analysed included pH, calcium (Ca), magnesium (Mg), sodium (Na), potassium (K), fluoride (F), chloride (Cl), nitrate (NO₃), sulphate (SO₄), calcium hardness (CaCO₃), phosphate (PO₄), magnesium hardness, total coliforms, faecal coliforms, and E. coli. For laboratory chemical analyses, sterile 500 mL bottles were utilized, while sterile 100 mL bottles were employed for microbiological analyses to collect water samples. All samples designated for laboratory analysis were properly labelled and stored in an icebox before transportation to the laboratory, where they were analysed within a 12-hour timeframe.

The collected samples underwent onsite and laboratory analysis. On-site, physical water quality parameters including EC, TDS, temperature, and pH were analysed using HANNA instruments. Figure 4.2 displays the HANNA multiparameter waterproof meter sets utilized for measuring physical parameters and dissolved oxygen on-site.



Figure 4.2 HANNA multiparameter waterproof meter set (HI98195) used to measure physical parameters and dissolved oxygen (Author, 2023)

Chemical and microbiological determinants were analysed at the laboratory of the Institute for Groundwater Studies at the University of the Free State where concentrations of calcium, magnesium, and calcium carbonate ions, were assessed by EDTA titration. Sodium and potassium ions were analysed through Atomic Absorption Spectroscopy whilst the ion-selective electrode was used to detect Fluoride concentration. The concentration of nitrate ions were detected through ion chromatography whilst the gravimetric analysis was used for detecting the sulphate ion concentrations. Microbial samples were tested using the Most Probable Number which involved inoculating water samples in multiple tubes and observing gas production to estimate bacterial counts.

Figure 4.3 depicts a flow diagram outlining the steps for water sampling at the study site.

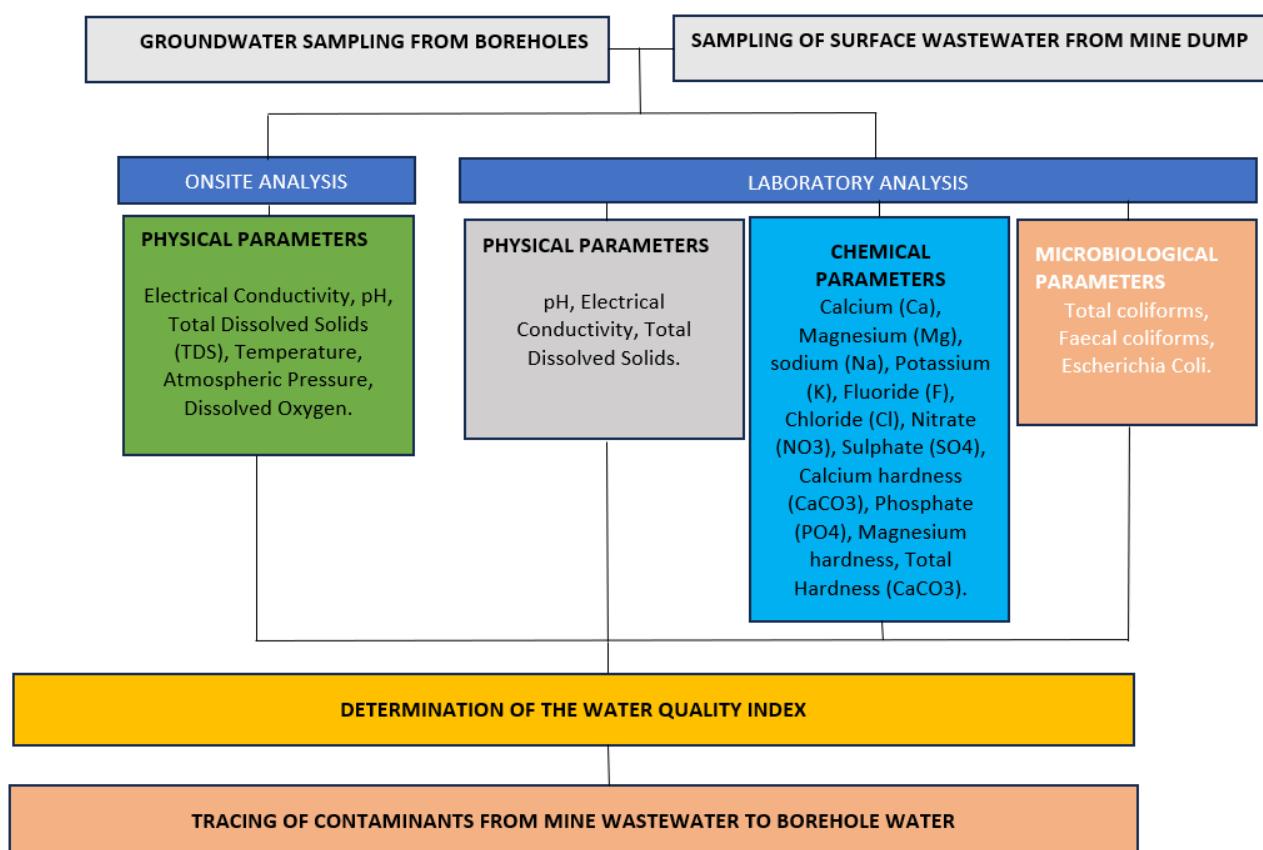


Figure 4.3 Flow diagram showing the steps for water sampling at the study site (Author, 2023)

The findings from the physical, chemical, and microbiological analyses were juxtaposed with the standards outlined in SANS 241 (2015) and WHO (2022) to determine the percentage compliance for each borehole sample and parameter. Furthermore, the water quality for each borehole and the stream water was evaluated using the weighted

arithmetic index method. Additionally, the microbiological parameters derived from the water samples were compared with the standards set forth in SANS 241 (2015). Figures 4.4 to 4.7 illustrate Boreholes 1 to 5 and the water sampling procedure for the stream effluent originating from the mine dump.



Figure 4.4 (a) Water sampling at Borehole 1; (b) Borehole 2; (c) 10 000 ℓ tanks that supply water from Borehole 2 to the students at the Welkom residence in the background



Figure 4.5 (a) Borehole 3; (b) Borehole 4; (c) Borehole 4 filter system seen at the inlet of the water into the 10 000 ℓ storage tank



Figure 4.6 Borehole 5

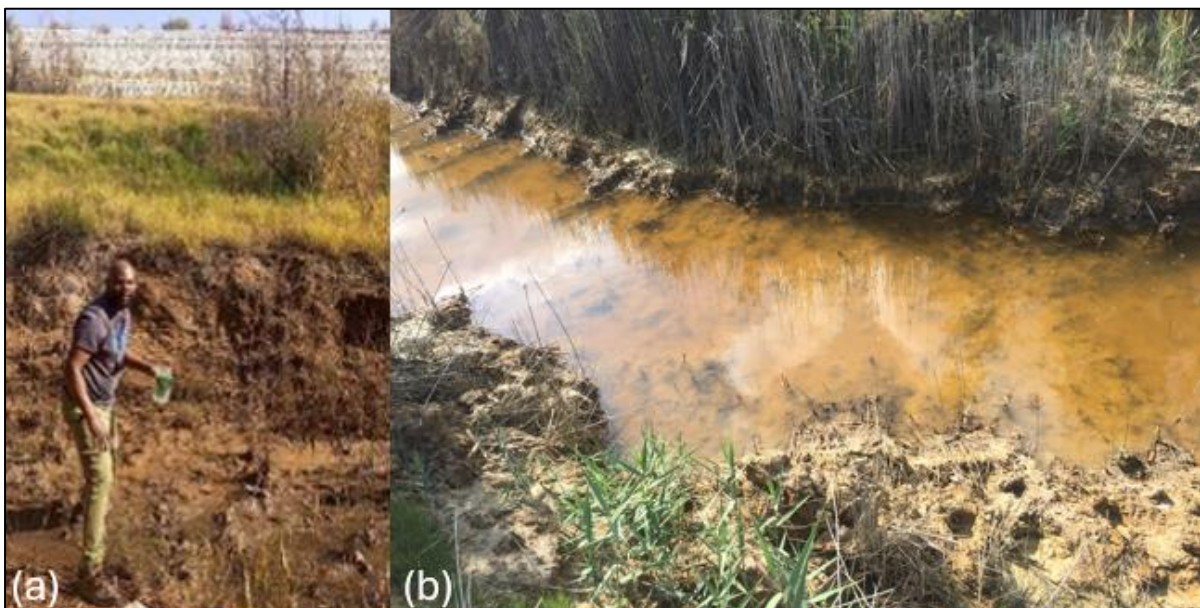


Figure 4.7 (a) Water sampling at mine dump effluent stream; and (b) mine dump effluent stream

4.2.2 Assessing the Water Quality Index

The Water Quality Index (WQI) stands out as an effective tool for summarizing water quality. Providing a straightforward and consistent measurement unit, it serves as a crucial aspect in the assessment and governance of groundwater and surface water. The primary objective behind employing the WQI is to establish a universal framework for consolidating all water quality parameters into a singular measurement unit that can be easily comprehended by individuals outside the water profession. Various methods for computing the WQI exist, with their suitability contingent on the specific contexts in which they are utilized. Often, the choice of WQI method is determined by the type of water under evaluation, which dictates the number of parameters considered in the

computation (Camejo et al., 2013). Nevertheless, a limitation of relying solely on the WQI is its inability to incorporate and analyse all parameters, potentially resulting in the oversight or omission of crucial data.

4.2.2.1 Selecting the methodology for calculating the Water Quality Index

Originating in the United States by Horton (1965), the Water Quality Index (WQI) has gained widespread adoption and acceptance across European, African, and Asian nations (Malan et al., 2003; Paun et al., 2016; Ritchie and Schiebe, 2000; Wertz and Shank, 2019). Renowned for its reliability, simplicity, and effectiveness, the WQI serves as a valuable tool for assessing water quality. It assesses water quality through a single value that is obtained by integrating the physical, chemical, and biological parameters under certain standard limits deemed to be relevant for a particular study. Since Horton (1965) proposed the first WQI, several arithmetic methods to calculate the WQI have been developed.

According to Lumb et al. (2011) and Paun et al. (2016), the development of the WQI was mainly based on four critical factors:

- The selection of water quality parameters suitable for the context of a particular study.
- Changing the raw parameter data into a common scale.
- Assigning relative weights to index components.
- Controlling the sampling design and aggregation of the water quality monitoring data.

The development of Horton's WQI paved the way for researchers to develop their own local indices. Several countries, including Britain, Canada, Malaysia and the United States, and international bodies such as the WHO, have designed their own indices, adapted to their own local specifications (Gorde and Jadhav, 2013; Paun et al., 2016; Strategic Assessment of Florida's Environment 1995; Sarkar and Abbasi 2006; Whitehead et al., 2009). Some of the most used WQI methods include the following:

- The Weighted Arithmetic Index method (WAI).
- The Florida Stream Water Quality Index.
- The National Sanitation Foundation Water Quality Index.

- The British Columbia Water Quality Index method.
- The Oregon Water Quality Index.
- The Canadian Council of Ministers of the Environment (CCME–WQI).

Each of these methods has its own scale for rating water quality depending on the intent and context of a particular study. Table 4.2 shows the ratings of the Weighted Arithmetic Index, the National Sanitation Foundation Water Quality Index, the Oregon Water Quality Index, and the Canadian Council of Ministers of the Environment methods as they are the mostly used in water quality assessments. Of all the WQI rating scales, the WAI has the most consistent scale for rating water quality with an equal spread of 25 units across all the five water quality bands. Furthermore, the WAI is the only water quality method that qualifies water ranking for both drinking and non-drinking purposes. These characteristics were critical for this study, as the groundwater vulnerability assessment to be developed should have the ability to give guidance on the suitability of the groundwater for drinking purposes.

Table 4.2 Ratings of different water quality indices

Weighted Arithmetic Water Quality Index (WAI) (1972)		National Sanitation Foundation Water Quality Index (1965)		Canadian Council of Ministers of the Environment (CCME-WQI) (2001)		Oregon Water Quality Index (1970)	
WQI value	Rating of water quality	WQI value	Rating of water quality	WQI value	Rating of water quality	WQI value	Rating of Water Quality
0–25	Excellent water quality	91–100	Excellent water quality	95–100	Excellent water quality	90–100	Excellent water quality
26–50	Good water quality	71–90	Good water quality	80–94	good Water quality	85–89	Good water quality
51–75	Poor water quality	51–70	Medium water quality	60–79	Fair water quality	80–84	Fair water quality
76–100	Very poor water quality	26–50	Bad water quality	45–59	Marginal water quality	60–79	Poor water quality
Above 100	Unsuitable for drinking purpose	0–25	Very bad water quality	0–44	Poor water quality	0–59	Very poor water quality

Table 4.3 shows the comparisons on four of the most popular groundwater vulnerability assessment methods. The characteristics of the four key WQI methods were further compared against an eight-scale criterion to determine the suitability of application of each of the WQI methods to the Welkom study area. Among the most considered

characteristics were the ease of application, calculation, and the flexibility of adding and removing parameters from the WQI method.

Table 4.3 Comparisons of the NSF (1965), CCME (2001), Oregon (1970) and WAI (1972) characteristics

Characteristics of the Water Quality Index method		NSF (1965)	CCME (2001)	Oregon (1970)	WAI (1972)
1	Includes physical properties	Yes	Yes	Yes	Yes
2	Includes chemical properties	Yes	Yes	Yes	Yes
3	Includes microbiological properties	Yes	Yes	No	Yes
4	Can be applied for different sampling rounds	No	Yes	Yes	No
5	Can be adapted to include additional parameters	Yes	No	Yes	Yes
6	Can be circulated using a single round of measurements	Yes	No	Yes	Yes
7	Ease of formula and calculation	No	No	Yes	Yes
8	Easy to interpret and analyse	No	Yes	No	Yes
Total “Yes” score		5	5	6	7

The WAI index method had a total “Yes” score of 7.0, compared to the other WQI methods such as the NSF, CCME, and the Oregon WQI, which had total scores of 5.0, 5.0, and 6.0 respectively. The WAI method had the main advantages:

- It can accommodate all the parameters in its assessments, including the physical, chemical, and microbiological.
- It can be adapted to include additional parameters.
- It is easy to calculate, analyse and interpret.

Moreover, the WAI method had the advantages in that it can be applied to the local South African context without any difficulty in interpretation. However, the major limitations to the application of the WAI method is that it can be difficult to apply to different sampling rounds that may be necessary for some long-term studies (Brown et al., 1973).

For this study, the WAI was chosen over the NSF (1965), CCME (2001), and Oregon (1970) methods. This is mainly because the WAI is less complicated compared to other WQI calculation methods. Additionally, the WAI method can be applied for single or multiple use measurements as the data is collected over a period of time (Brown et al., 1973).

4.2.3 Weighted Arithmetic Index method

Given the varying time intervals and frequencies of water sample collection throughout the season, the Weighted Arithmetic Index (WAI) method was chosen as the most appropriate approach for determining water quality. Additionally, the WAI employs simpler calculations, making it easier to implement compared to other methods used for calculating the Water Quality Index (WQI).

The WAI can be calculated through the following steps:

Step 1: Calculation of the unit weight (W_i) for each water quality parameter to the recommended standard for the corresponding parameter

The unit weight values were calculated by using the following formula as recommended by Tiwari and Misha (1985, cited in Gorde and Jadhav, 2013):

$$W_i = K \sum_{n=1}^n \frac{1}{S_i} \quad \text{Equation 4.1}$$

Where, S_i is the recommended standard for the corresponding i^{th} parameter.

The proportionality constant K is obtained by using the following formula:

$$K = \frac{1}{\sum_1^n \frac{1}{S_n}} \quad \text{Equation 4.4}$$

Step 2: Calculation of the quality rating scale (Q_i) for each parameter was conducted by using the expression:

$$Q_i = 100 \left[\frac{Q_{\text{actual}} - Q_{\text{ideal}}}{S_{\text{standard}} - Q_{\text{ideal}}} \right] \quad \text{Equation 4.3}$$

Where,

- Q_{actual} = actual or estimated amount of the i^{th} parameter
- Q_{ideal} = the ideal value of the parameter
- S_{standard} = recommended SANS 241 standard for the corresponding parameter
- Q_{ideal} = 0 (except for pH = 7, and dissolved oxygen = 14.6 mg/l⁻¹)

Step 3: The overall WQI was calculated using the following equation:

$$\text{Weighted Average WQI} = \frac{\sum_{i=1}^{i=n} Q_i W_i}{\sum w_i} \quad \text{Equation 4.4}$$

The standard rating scale for the WAI is shown in Table 4.4.

Table 4.4 Weighted Arithmetic Index rating scale

Water quality index	Water quality class
0–25	Excellent
26–50	Good
51–75	Poor
76–100	Very poor
Above 100	Not suitable for drinking purposes

Source: Ratikane (2013); Rickwood and Carr (2007)

4.2.4 Results of the WQI for groundwater near mine dumps in Welkom

Upon comparing the average physical and chemical parameters with the drinking water quality guidelines outlined in SANS 241 (2015) and WHO (2022), it was observed that the water quality parameters in both the boreholes and the mine stream exceeded the recommended limits. Table 4.5 illustrates the percentage compliance of each borehole (B1–B5) and the stream water (S6) with respect to the SANS 241 and WHO (2022) water quality guidelines. The results are average values from the data collected in the summer and autumn seasons.

Table 4.5 Comparison of the physical and chemical determinants with the SANS 241 and WHO limit standards

Parameter	Limit standard (SANS 241)	Limit standard (WHO)	Risk	B1	B2	B3	B4	B5	Mean for boreholes	S6 (open stream)	Minimum	Maximum
EC (Electrical Conductivity)	(170 mS/m)	-	Acute Health	84.62	<14.7	202.31	62.75	138.89	122.14	2 051.42	<14.7	2 051.42
TDS (Total Dissolved Solids)	(1 200 mg/ℓ)	1000	Aesthetic	579.77	85.97	1 234.44	433.16	891.73	645.01	17 531.61	85.97	17 531.61
pH	(5-9.7)	8.5	Operational	7.04	7.55	6.80	7.64	6.81	7.17	7.17	6.80	7.64
Calcium (Ca)	(150 mg/ℓ)	75	Aesthetic	68.60	10.82	79.63	55.61	93.06	61.54	896.04	10.82	896.04
Magnesium (Mg)	(70 mg/ℓ)	30	Aesthetic	28.69	3.21	41.37	22.98	36.53	26.56	856.49	3.21	41.37
Sodium (Na)	(200 mg/ℓ)	200	Aesthetic	73.92	13.22	307.61	49.72	146.79	118.25	3 170.74	13.22	3 170.74
Potassium (K)	(50 mg/ℓ)	10	Aesthetic	12.67	4.30	29.20	8.25	25.47	15.98	304.65	4.30	29.20
Fluoride (F)	(1.5 mg/ℓ)	1.5*	Chronic Health	0.37	<0.10	0.39	0.30	0.35	0.35	1.89	<0.10	1.89
Chloride (Cl)	(300 mg/ℓ)	250*	Aesthetic	85.25	18.89	317.05	63.82	194.54	135.91	6197	18.89	6 197
Nitrate (NO ₃ ⁻)	(11 mg/ℓ)	50*	Acute Health	1.76	0.36	0.29	2.00	4.16	1.71	0.75	0.36	4.16
Sulphate (SO ₄)	(500 mg/L)	250	Acute Health	134.85	1.11	146.51	99.40	89.36	94.25	3 075.46	1.11	3 075.46
Total Hardness (CaCO ₃)	(100 mg/L)	-	Aesthetic	289.65	40.25	369.21	233.49	382.81	263.1	5 764.44	40.25	5 764.44
% Compliance				92	100	58	92	92	100	17		

Key:	Sample exceeds the SANS (2015) limit standard	
	WHO Guidelines for drinking water quality (2022); standards of health concern	*

Among the six samples analysed, samples B3 and S6 exhibited the lowest compliance rates. Sample B3 surpassed the recommended limits for EC and TDS physical parameters by approximately 19% and 3%, respectively, while the stream water samples exceeded these parameters by more than 200% and 14%, respectively.

Regarding chemical parameters, borehole samples B1, B3, B4, and B5 exceeded the recommended guidelines for total hardness attributed to CaCO_3 according to both SANS 241 and WHO (2022). Furthermore, Sample B3 displayed elevated concentrations of Na and Cl.

Stream water sample S6 surpassed all chemical water quality guidelines except for pH and NO_3^- . With a compliance percentage of only 17%, the S6 sample exhibited elevated levels of F and SO_4 , which are known to contribute to various chronic and acute human diseases.

Several significant water quality parameters concerning aesthetics and health were graphed against the SANS 241 limit standards. Figures 4.8 to 4.10 depict the plots of EC, TDS, SO_4 , NO_3 , F, and Na^+ parameters against the SANS 241 limit standards.

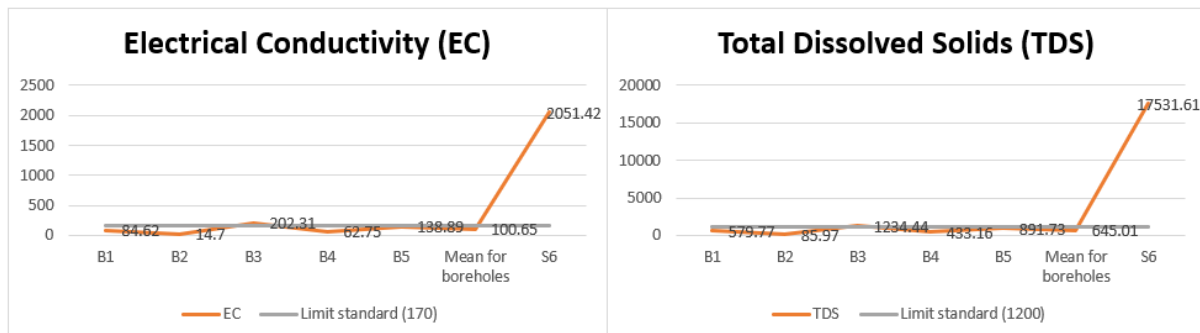


Figure 4.8 Mean electrical conductivity and total dissolved solids values against their limit standards

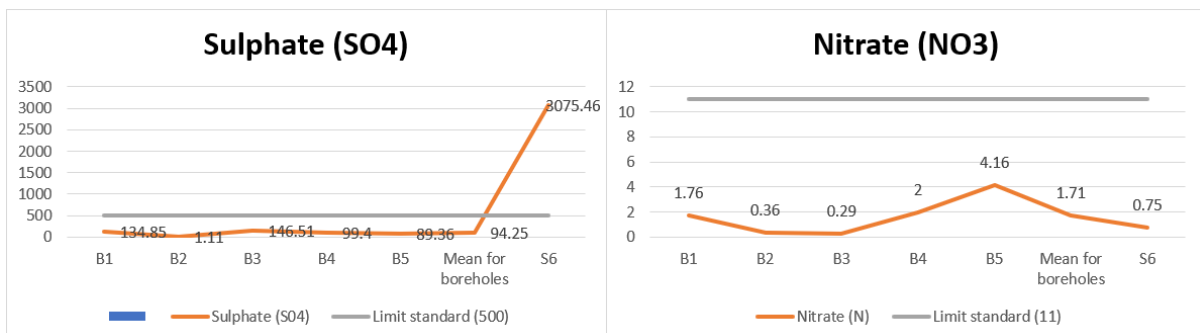


Figure 4.9 Mean sulphate and nitrate values against their limit standards

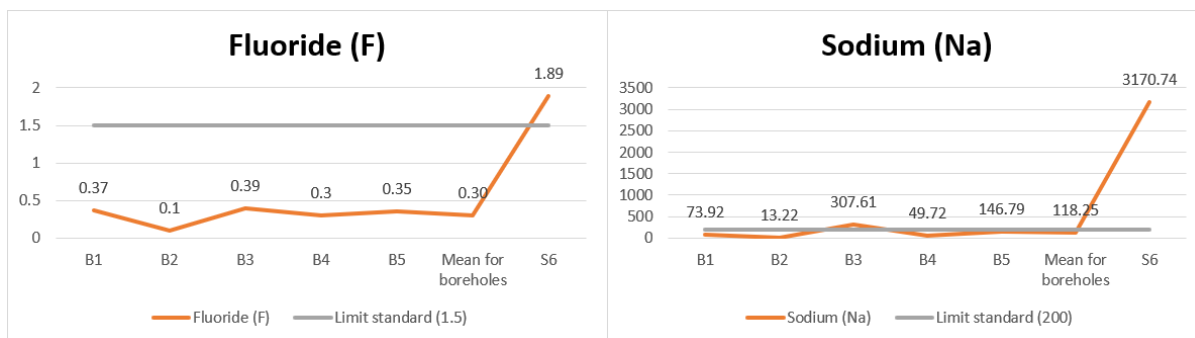


Figure 4.10 Mean fluoride and sodium values against their limit standards

While most of the parameters tested did not exceed their limit standards for the borehole water samples, almost all the parameters exceeded their limit standards for the mine dump sample S6. The high conductivity values, coupled with the high TDS concentrations, can be an indication of the high levels of contaminant loads within the mine dumps. Furthermore, the high levels of SO_4 and NO_3 concentrations indicated high levels of contamination due to AMD.

4.2.5 Comparison of the microbiological parameters with the SANS limit standards

The microbial parameters were analysed by assessing total coliforms, faecal coliforms, and *E. coli* in both borehole and stream samples during the summer and autumn seasons of 2022, following the SANS 241 (2015) standards. During the summer, samples B2, B3, and B5 from boreholes showed elevated levels of total and faecal coliforms, while sample S6 surpassed all SANS 241 limits for the parameters studied. Table 4.6 illustrates the percentage compliance of microbiological parameters with SANS 241 standards for the summer season.

In the autumn season, all five borehole samples and the stream sample exceeded the SANS 241 standard for total coliforms. However, none of the samples exceeded the standards for faecal coliforms, and all but B3 exceeded the *E. coli* SANS standard, as detailed in Table 4.7.

Table 4.6 Comparison of microbiological parameters-Summer Season

Parameter	SANS 241 Standard	Risk	B1	B2	B3	B4	B5	S6
Total coliforms	(<10 counts/100 ml)	Operational	2	>2 420	>2 420	<1	127	>2 420
Faecal coliforms	(Not detected in 100 ml)	Acute Health	<1	67	4	<1	<1	613
<i>E. coli</i>	(Not detected in 100 ml)	Acute Health	<1	<1	<1	<1	<1	1
Compliance %				33	33	100	67	0

Key Sample exceeds the SANS (2015) limit standard

Table 4.7 Comparison of microbiological parameters-Autumn Season

Parameter	SANS 241 Standard	Risk	B1	B2	B3	B4	B5	S6
Total coliforms	(<10 counts/100 ml)	Operational	>2 420	>2 420	10	2 420	>1 000	>2 420
Faecal coliforms	(Not detected in 100 ml)	Acute Health	326	1 986	1	112	816	613
<i>E. coli</i>	(Not detected in 100 ml)	Acute Health	5	>2 420	<1	1	6	1
Compliance %				33	66	33	33	33

Key Sample exceeds the SANS (2015) limit standard

Table 4.8 displays the WQI for the six samples alongside their respective water quality classifications. The WQI values were computed utilizing the WAI method, employing both the SANS 241 and WHO (2022) water quality guidelines. Among the five boreholes, three exhibited poor water quality, while the remaining two demonstrated good water quality. The average WQI derived from all boreholes, considering the physical and chemical factors, indicated an overall poor water quality.

Table 4.8 Water Quality Index

Borehole sample	Water Quality Index	Water quality class
B1	64.26	Poor
B2	44.79	Good
B3	63.06	Poor
B4	31.78	Good
B5	51.06	Poor
Mean WQI (B1–B5)	68.16	Poor
Stream	544.31	Not suitable for drinking purposes

The findings derived from the WQI computed using the WAI method, employing both the SANS 241 and WHO (2022) water quality guidelines, revealed that three out of the five boreholes exhibited poor water quality, whereas only two boreholes demonstrated good water quality. Moreover, the average WQI, calculated based on physical and chemical parameters, also indicated poor water quality overall.

In addition, the WQI for the stream flowing through the mine dump effluent measured 544.31, which surpassed the minimum guideline requirement of 100 for very poor water quality by fivefold. Consequently, the water was deemed unsuitable for drinking purposes.

4.2.6 Discussion and Conclusion

The study findings suggest that the groundwater quality from the boreholes situated at the CUT campus cannot be deemed safe for drinking purposes. Even though the water quality indices may imply suitability for drinking for some of the boreholes, it remains unfit for human consumption due to microbiological water quality parameters surpassing the thresholds outlined in the drinking water quality guidelines established by both SANS 241 and WHO (2022) standards.

The elevated levels of contaminants observed in the stream adjacent to the mine dump may likely result from significant leaching of pollutants from the dump. Likewise, the higher-than-average contamination of sample B3 could be associated with its close proximity to the contaminated stream water. However, no consistent direct correlations were identified between determinant concentrations in any sample and the presence of microbial organisms across both seasons. To delve deeper into potential correlations with microbial organisms, additional WQI assessments involving heavy metals may be warranted.

Microbial contamination levels in the groundwater exhibited significant variation between the autumn and summer seasons. During autumn, total coliforms and *E. coli* were more prevalent, whereas total coliforms and faecal coliforms dominated during summer. These seasonal shifts in microbial composition likely stem from the distinct weather patterns characteristic of each season, which contribute differently to microbial proliferation. Given these fluctuations, it's imperative to closely monitor seasonal changes in microbial growth, as they may pose unforeseen health risks to the Welkom population.

The groundwater in the CUT Welkom campus vicinity has been significantly contaminated by various physical, chemical, and microbiological factors originating from the nearby mine dump. This contamination has notably compromised the water quality, rendering it unsuitable for drinking. While some borehole water samples met certain criteria outlined in the SANS 214 and WHO (2022) water quality guidelines for specific determinants, none of the samples complied fully with all parameters. Consequently, utilizing this groundwater for drinking purposes could potentially lead to adverse health effects in humans. Further testing is essential to evaluate the presence of heavy metals or trace elements in the groundwater, thereby assessing its susceptibility to contamination from the mine dump.

4.3 Characterisation of aquifer systems in the study area: Identifying aquifer type, detecting fracture points, and groundwater flow direction

4.3.1 Introduction

In general terms, aquifers are referred to as bodies of saturated rocks or geological formations through which water find its way into wells and springs (ASTM International,

2012). The movement of water in aquifers mainly depends on the permeability of the aquifer material. Permeable material is composed of a network of interconnected cracks that allow for spaces in which water can move freely. As the permeability of several materials differ, so does the rate at which they allow for contaminant movements from the sources. Depending on the location and environment, some permeable materials may allow groundwater to move for several meters a day, while some less permeable materials may allow groundwater to move only several centimetres in decades. Studies have shown that groundwater moves very slowly through relatively impermeable materials such as clay and shale, while it moves relatively fast in sandy and loam soils (Dong et al., 2020). The determination of the aquifer type in this study, as well as its characterisation, is thus essential in investigating groundwater vulnerability to contamination. Figure 4.11 shows a simplified general aquifer development process.

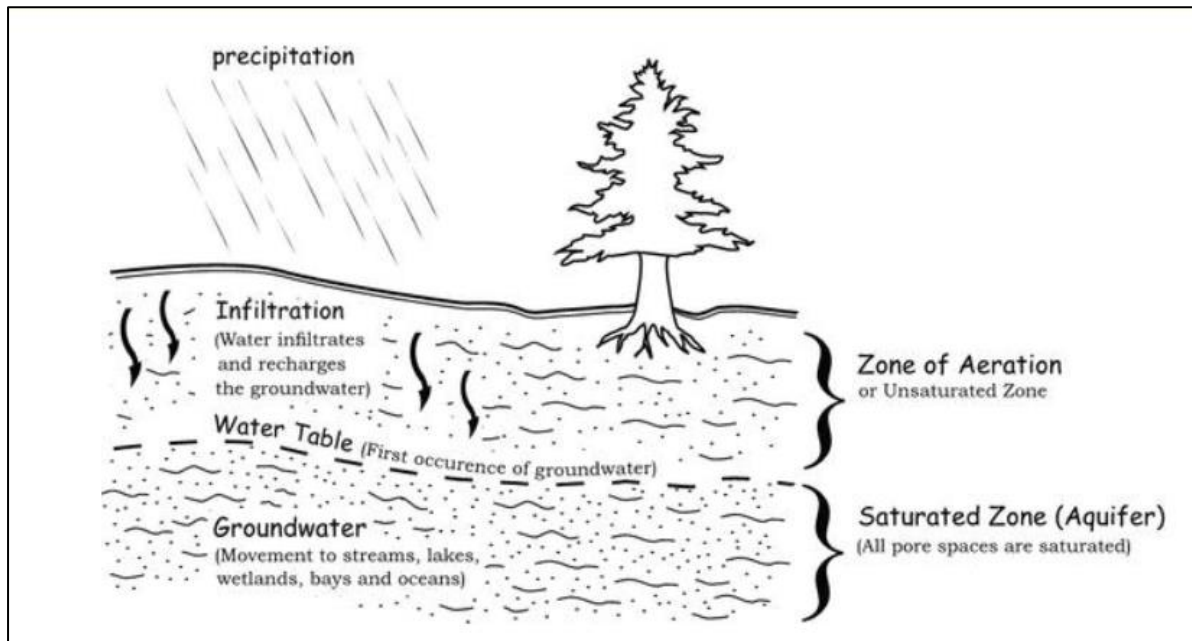


Figure 4.11 *Aquifer development process* (Salako and Adepelumi, 2018)

Aquifers can be classified as either confined or unconfined depending on their formation and characteristics. Confined aquifers are those bodies of water found accumulating in a permeable rock and are usually enclosed by two impermeable rock layers or rock bodies, while unconfined aquifers are generally found near the land surface and have no layers of clay or other impermeable geologic material. The classification of aquifers is dependent on the water table location, hydraulic conductivities, as well as its structure. Figure 4.12 shows the classification and structure of confined and unconfined aquifers.

Other petrophysical properties such as permeability, porosity, and seismic velocities of the subsurface can also contribute to the characterisation of the aquifer.

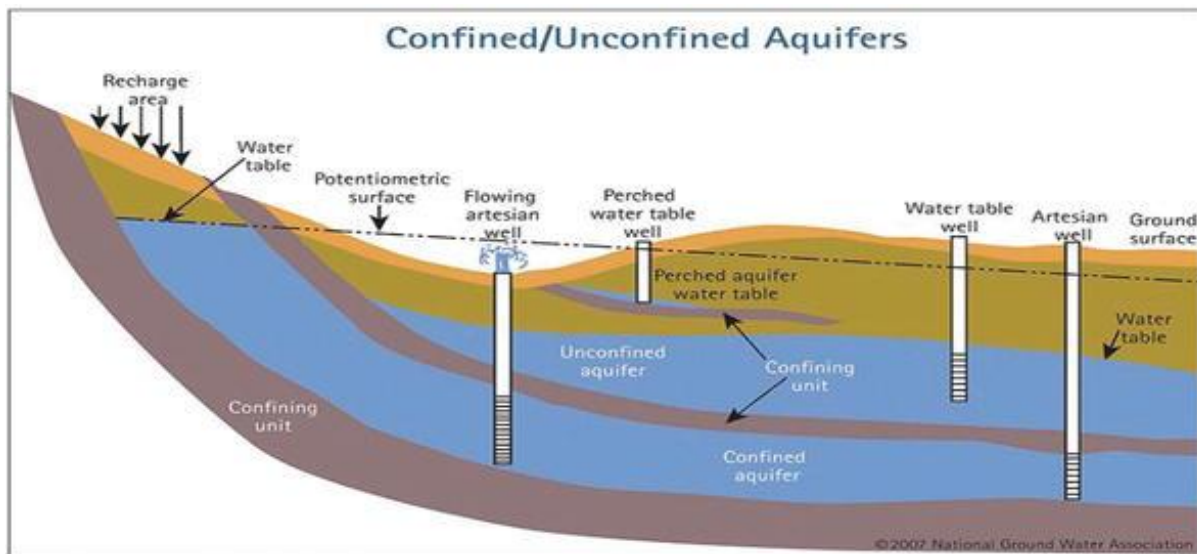


Figure 4.12 Classification of confined and unconfined aquifers (NGWA, 1999)

The characterisation of an aquifer can be conducted by using geophysical techniques such as electrical resistivity, electromagnetic induction, ground penetrating radar and seismic techniques (Salako and Adepelumi, 2018). This study was aimed at identifying and classifying the aquifer types in the Welkom area using the electrical resistivity method and a set of observed characteristics. The transmission of contaminants from the mine dump to the boreholes is largely influenced by the type of aquifer. The assessment of the groundwater vulnerability to contamination in the Welkom area is therefore linked to the aquifer type. Table 4.9 shows the comparison of various hydrogeologic properties for granular, fractured rock, and karst aquifers.

Table 4.9 Comparison of various hydrogeologic properties for granular, fractured rock, and karst aquifers

Aquifer characteristics	Aquifer type		
	Granular	Fractured rock	Karst
Effective porosity	Mostly primarily through intergranular pores	Mostly secondary through joints, fractures, and bedding plane partings	Mostly tertiary (secondary porosity modified by dissolution): through pores, bedding, planes, fractures, conduits, and caves
Isotropy	More isotropic	Probably anisotropic	Highly anisotropic
Homogeneity	More homogeneous	Less homogeneous	Non-homogeneous
Flow	Slow, laminar	Possibly rapid and possibly turbulent	Likely rapid and likely turbulent

Aquifer characteristics	Aquifer type		
	Granular	Fractured rock	Karst
Flow predictions	Darcy's law usually applies	Darcy's law may not apply	Darcy's law rarely applies
Storage	Within saturated zone	Within saturated zone	Within both saturated zone and epikarst
Recharge	Dispersed	Primarily dispersed with some point recharge	Ranges from almost completely dispersed to almost completely point recharge
Temporal head variation	Minimal variation	Moderate variation	Moderate to extreme variation
Temporal water chemistry variation	Minimal variation	Minimal to moderate variation	Moderate to extreme variation

Source: ASTM International (2012)

4.3.2 Evaluation of aquifer fracture zones and groundwater flow direction

4.3.2.1 Introduction

Identifying and studying aquifer characteristics such as fracture zones, groundwater levels, and groundwater flow direction is essential in mapping the vulnerability of groundwater to contamination. According to Kirsch (2006), the detection of fracture zones is an important aspect of hydrogeological explorations as they play a huge role in groundwater supply. This study attempted to identify and locate the aquifer type, fracture zones from the boreholes, and mine dump effluent in the study area at the CUT Welkom campus. The five boreholes are in very close proximity to mine dump deposits located between 270 E and 270 S. As discussed in the previous chapter, the geology of the Welkom area is largely classified under the Witwatersrand Supergroup, which is part of the Karoo Supergroup and the Ventersdorp Supergroup (McCarthy et al., 2018). The Karoo bedrock strata are generally massive, with very low porosity, except for that provided by occasional fractures. According to Frimmel and Minter (2002), the rocks characterising the Welkom area are mainly composed of calcite, shale, and quartzite. This means that the movement of groundwater within the study area could be largely due to fracture points, which this study attempted to identify.

4.3.2.2 Methodology

According to Love et al. (2002), it's suggested that the natural stratification of EC within a borehole can mirror that of the aquifer, particularly in fractured aquifers. This stratification may serve as a useful indicator to identify potential locations of primary

conductive points within the aquifer, which facilitate the flow of water towards the borehole. Hydrological observations, coupled with sub-surface investigations involving EC and resistivity, were used to assess the fracture zones and groundwater flow direction. The electrical methods were chosen because they are easy to apply and involve the use of less complicated calculations as compared to the traditional drilling and logging methods (Dong et al., 2020). Furthermore, the electrical resistivity method is a commonly used and reliable method for detecting fracture zones through the assessment of horizontal variations in electrical resistivity for sedimentary basins (Dakir et al., 2019).

The EC and resistivity measurements obtained from the five boreholes were used to infer lithological information and hydrogeological parameters needed for mapping fracture zones and groundwater flow direction. EC techniques take into account uncased portions of a borehole that intersect multiple active fractures or conductive zones. Under normal conditions, borehole salinity is anticipated to progressively rise with increasing borehole depth, with EC remaining constant in stagnant sections but fluctuating in flowing sections. In areas of flow, water quality is influenced by the characteristics of water from inflow points or permeable fracture layers, whereas in stagnant sections, water quality is primarily determined by the quality of water stored in the formation, influenced by the water's residence time.

4.3.2.3 Measuring the depth to groundwater

Groundwater level observations were conducted at the five boreholes to assess groundwater levels within the study area. The depth of the groundwater level in each borehole was determined using a Solinst rolling meter, measured from the groundwater level to the ground surface. For each measurement, the Solinst meter electrode was inserted into the borehole to determine both the groundwater level and the borehole depth. These measurements were conducted during the December months in summer and April months in autumn seasons respectively, spanning from 2022 to 2024. Figure 4.13 illustrates the Solinst meter utilized for groundwater level measurements.



Figure 4.13 Solinst meter used for calculating groundwater level (Author, 2023)

Table 4.10 shows the recorded values of the depth to groundwater, depth of borehole as well as the calculated range of values for each of the five boreholes. Out of the five boreholes, Borehole 2 had the lowest depth to groundwater while Borehole 3 had the highest depth to groundwater. An assessment of the boreholes ranges also indicated that Boreholes 3 and 1, had the highest and lowest ranges respectively. This presents the likelihood that groundwater flows in the direction of Borehole 2, to Borehole 5, to Borehole 4 and then to Borehole 3 although this still needs to be confirmed with the relevant tests. The low values obtained in the depth to groundwater together with the reduced ranges and borehole depths might be an indication of the existence of a shallow aquifer which is supplying all the boreholes in the study area.

Table 4.10 Range and depth to groundwater of the five boreholes at the study site

Parameter	Borehole				
	B1	B2	B3	B4	B5
Depth to groundwater (m)	2.30	1.87	2.71	2.44	2.30
Depth of borehole (m)	18.0	31.7	47.2	25.54	23.2
Range (Depth of borehole – Depth to groundwater) (m)	15.7	29.8	44.49	23.1	20.9

4.3.2.4 Assessing the electrical conductivity profiling and fracture points of the aquifer using the dilution test

To determine the EC profiling of the aquifer, the electrical conductance of each of the five boreholes was measured at 1 m intervals using the HANNA-HI98195 instrument. Following that, a dilution test was used for assessing the existence and location of fracture points for all five boreholes. Common salt (sodium chloride) was used as the natural conductive tracer for each of the boreholes. During the assessment, the salt was tied in a porous sock, which was tied to a rope. The sock was inserted into each of the

boreholes at intervals of 1 m depths, and the electrical conductance and resistivity values of the boreholes were recorded at each depth using the HANNA-HI98195 instrument. The areas of the least electrical conductance (high resistivity) indicated the possibility of fracture zones in each of the borehole depths at varying points. The delineation of connected pore spaces, and fractures within the underlying rocks filled with water, led to the variations in conductivity and resistivity values. Furthermore, the distribution of conductance or resistivity obtained from the measurements also revealed the location of electrically conductive pathways between boreholes, which correlated with the cavities.

The EC values obtained from the EC and dilution test are shown in Table 4.11. The EC under saline conditions represents the conductivity values with salt inserted, while the EC under non-saline conditions represent the conductivity values of groundwater before the salt was inserted. The conductivity values for each of the five boreholes were recorded at 1 m intervals. The conductivity values highlighted in red indicate the lowest recorded values for each borehole at the insertion of salt across the borehole depth.

Table 4.11 Electrical conductivity values at various borehole depths from the dilution test

Depth (m)	Borehole (Electrical conductivity values mS/m)									
	Non-Saline	Saline	Non-Saline	Saline	Non-Saline	Saline	Non-Saline	Saline	Non-Saline	Saline
	B1		B2		B3		B4		B5	
1	621	1 517	134	1 533	1 396	1 978	627	1 020	623	1 240
2	713	1 520	156	1 548	1 381	1 970	635	1 023	654	1 222
3	619	1 430	230	1 628	1 372	1 966	571	1 016	683	1 194
4	647	1 530	122	1 630	1 420	1 988	652	1 029	627	1 201
5	466	1 596	154	1 605	1 423	1 984	531	1 032	621	1 191
6	411	1 660	166	1 578	1 477	1 843	577	1 028	656	1 201
7	443	1 540	311	1 640	1 422	1 780	429	994	692	1 142
8	682	1 604	252	1 200	1 450	1 738	562	986	643	1 182
9	691	1 361	156	987	1 469	1 716	573	964	673	1 176
10	595	875	146	604	1 472	1 704	581	978	622	874
11	721	980	172	744	1 486	1 706	667	960	680	1 180
12	659	1 221	167	629	1 434	1 698	621	952	762	1 182
13	652	1 278	272	988	1 465	1 704	627	954	762	1 180
14	721	1 312	144	1 260	1 450	1 702	635	950	652	1 186
15	678	1 271	456	1 330	1 452	1 701	571	956	472	1 181
16	801	1 281	652	1 489	1 403	1 770	652	942	682	1 278
17	762	1 292	562	1 403	1 420	1 780	531	942	528	1 298

Depth (m)	Borehole (Electrical conductivity values mS/m)									
	Non-Saline	Saline	Non-Saline	Saline	Non-Saline	Saline	Non-Saline	Saline	Non-Saline	Saline
	B1		B2		B3		B4		B5	
18	652	1 322	561	1 589	1 451	1 760	577	946	688	1 303
19	467	1 456	489	1 693	1 483	1 744	429	940	568	1 372
20			622	1 760	1 401	1 250	562	946	629	1 374
21			592	1 777	1 496	1 720	573	949		
22			480	1 773	1 443	1 728	581	948		
23			650	1 740	1 420	1 720	667	848		
24			454	1 699	1 396	1 718	621	738		
25			540	1 776	1 381	1 711				
26			620	1 788	1 372	1 748				
27			583	1 793	1 420	1 733				
28			658	1 754	1 423	1 704				
29			681	1 788	1 477	1 782				
30			871	1 790	1 672	1 786				
31					1 562	1 988				
32					1 492	1 872				
33					1 492	1 602				
34					1 562	1 732				
35					1 346	1 811				
36					1 462	1 804				
37					1 589	1 759				
38					1 389	1 793				
39					1 482	1 829				
40					1 572	1 850				
41					1 492	1 851				
42					1 367	1 829				
43					1 692	1 822				
44					1 572	1 839				
45					1 423	1 854				

Graphs showing the EC profile and the dilution test profile indicating the fracture points were plotted against the borehole depths, and the results are shown in Figure 4.14. The points showing sudden dips in EC under saline conditions represent the fracture points. A detailed description of each of the boreholes is given below:

Borehole 1

The lowest recorded EC values for Borehole 1 were at 875 mS/m at a depth of 10 m under saline conditions. The EC values under non-saline conditions ranged between 411 mS/m

and 801 mS/m, while it ranged between 875 mS/m and 1 788 mS/m under saline conditions.



Figure 4.14 Graphs showing electrical conductivity values in saline and non-saline environments for boreholes B1–B5

Borehole 2

Three significant EC drops were identified for Borehole 2 under saline conditions. The drops occurred at the depths of 10 m, 12 m, and 24 m, giving a possibility of the existence of three fractures. The electrical conductance for the borehole under non-saline conditions remained stable with the lowest conductance of 134 mS/m at a depth of 1 m and the highest conductance of 871 mS/m happening at a depth of 30 m.

Borehole 3

Borehole 3 had a drop in EC of 1 250 mS/m and 1 428 mS/m at depths of 18 m and 22 m under saline conditions, respectively, while the EC values for non-saline conditions ranged from 1 372 mS/m to 1 672 mS/m, giving a possibility of two fractures.

Borehole 4

Borehole 4 had a sharp decline in EC of 738 mS/m at a depth of approximately 24 m under saline conditions, while a conductivity of 950 mS/m was recorded at a depth of 14 m under non-saline conditions. The EC values under non-saline conditions ranged between 429 mS/m and 667 mS/m, while the range under saline conditions was between 738 mS/m and 1 029 mS/m.

Borehole 5

The lowest EC values for Borehole 5 under saline conditions were 874 mS/m and 1 142 mS/m and occurred at the depths of 10 m and 7 m, respectively. The range of EC values were between 1 114 mS/m and 1 687 mS/m for saline and 621 mS/m and 692 mS/m under non-saline conditions.

The fracture points and depths to groundwater values across the five boreholes are shown in Figure 4.15.

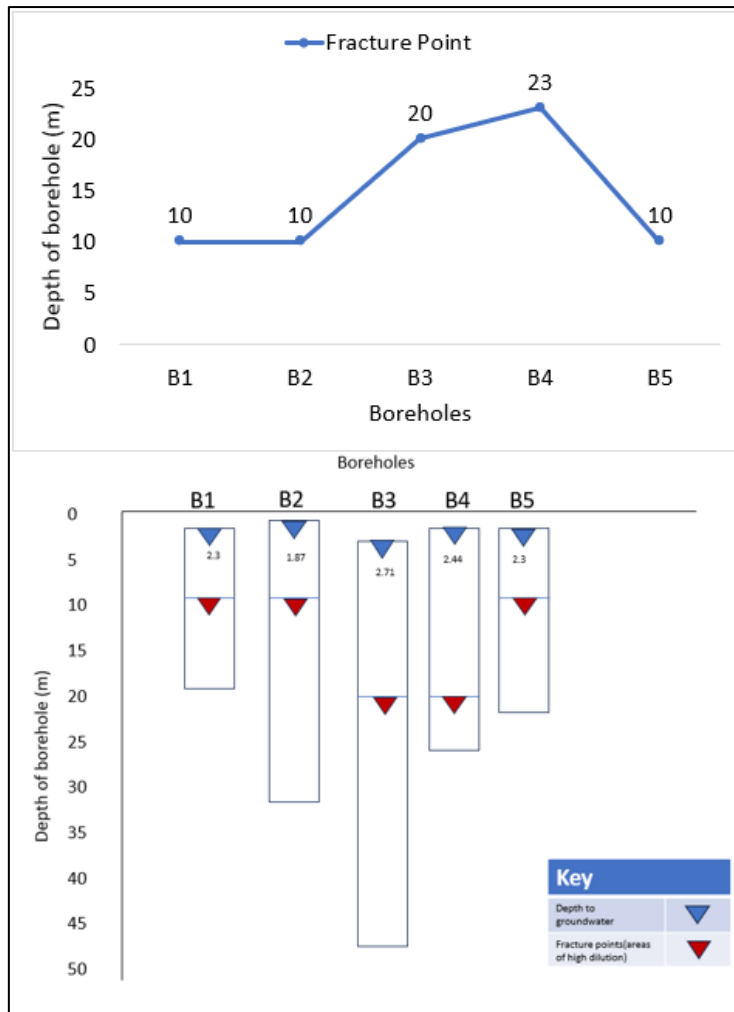


Figure 4.15 Borehole depths against fracture points for the five boreholes in the study area

Boreholes 1, 2, and 5 had fractures showing at the same depths of 10 m, while Boreholes 3 and 4 were showing fractures at approximately 20 m. The occurrence of fracture points consistently between Boreholes 1, 2 and 5 and Boreholes 3 and 5 is a possible indication on the existence of an unconfined fractured aquifer within the study area. This is a huge risk to groundwater contamination as groundwater in shallow unconfined aquifers can be easily contaminated.

The data from the EC and dilution tests conducted over the five boreholes indicated the presence of shallow unconfined fracture zones. This suggests that all five boreholes in the study may be supplied by a shallow unconfined aquifer.

4.3.2.5 Assessing the groundwater flow direction

The depth to groundwater values were used to identify the groundwater flow direction at the study site. Using boreholes B2, B3, and B5 as the anchor points, equipotential lines were plotted to identify the direction of groundwater flow using the following steps:

Step 1

Boreholes B2, B3, and B5 were selected among the five boreholes in the study site and the depth to groundwater for each borehole was noted as 1.9 m, 2.7 m, and 2.3 m respectively. The choice of selecting the three boreholes was made based on the following assumptions:

- The distance between the boreholes covers a large portion of the study area.
- The boreholes belong to the same aquifer system as indicated by the data from the EC and dilution tests.

Step 2

- Each of the three wells were joined together and the space between the wells was divided into segments to interpolate the values of head between the wells.
- The space between the wells was divided into segments to linearly interpolate the values of the heads between the wells, as shown in Figure 4.16.



Figure 4.16 Connection of head values between Boreholes B2, B3, and B5 (Google Maps, 2023)

Step 3

- Dashed equipotential lines were drawn to connect equal values of interpolated heads.
- The aquifer rock type was assumed to have isotropic and homogeneous hydraulic conductivity, and the groundwater flow occurred in the direction of maximum gradient which is at right angles to the equipotential lines.
- The overall groundwater flow direction was detected to be in the north-westerly direction from boreholes B1 and B2. Figure 4.17 shows the proposed groundwater flow direction in relation to the borehole location as well as the mine dump.

The analysis of the groundwater flow direction data, indicates that groundwater flows predominantly in a north-westerly direction from the boreholes. This pattern of flow presents a lower likelihood of contamination from the nearby mine dumps, as the flow appears to be carrying groundwater away from these sources. The assessment, therefore, reduces the immediate concern that elements from the nearby mine dumps are contaminating the groundwater at the borehole sites. However, the absence of compelling evidence linking contamination in the boreholes to the nearby mine dump raises two key possibilities.

Firstly, it is plausible that other mine dumps in the surrounding area, which have not been the focus of the investigation, may be contributing to the contamination of the groundwater in the region. Groundwater is influenced by various environmental and geological factors, including hydraulic gradients, subsurface lithology, and anthropogenic activities. As such, these factors might be directing contamination from more distant or overlooked sources into the boreholes. It is important, therefore, to widen the scope of investigation to consider the cumulative impact of multiple mine dumps in the area, especially those situated along potential alternative flow paths.

Secondly, the data may indicate a potential shift in groundwater flow direction as the groundwater approaches the boreholes. Subtle variations in the subsurface geology, such as changes in soil permeability or bedrock composition, could influence the local hydraulic gradients, resulting in deviations from the broader regional flow pattern. If this is the case, the groundwater could be rerouted in the vicinity of the boreholes, bringing contamination from nearby mine dumps despite the overall flow direction pointing away

from them. A more detailed hydrogeological survey, including seasonal fluctuations and localized flow dynamics, could help clarify whether this is occurring.



Figure 4.17 (a) Proposed groundwater flow direction across the five boreholes, (b) Groundwater flow direction relative to the mine dump (Google Maps, 2023)

The validation of the groundwater-flow direction conducted through a comparison of the physical and chemical water quality parameters for the five boreholes against the SANS 241 water quality standard guidelines indicated that borehole B3 had five parameters of EC, TDS, Na, Cl, and CaCO₃ exceeding the recommended SANS standard limit as

compared to the other boreholes B1, B3, B4, and B5, which only exceeded the limits in total water hardness (refer to Table 4.5). This can be linked to the fact that the position of borehole B3 was downstream, and hence, receives all the contaminant residues from other boreholes as well as from the mine dump as indicated in the ground flow direction mapping.

4.4 Conclusion

The study outcomes indicates that the fracture zones for all the boreholes within the study area occurred at depths between 9 m and 24 m. Except for Boreholes 3 and 4, which had their depths at approximately 20 m, Boreholes 1, 2, and 5 had their fracture points detected at approximately 10 m depths from the ground level. The availability of fracture zones at shallow depths for the boreholes can be an indication that the boreholes are situated in a shallow unconfined aquifer. The presence of a shallow unconfined aquifer supplying the boreholes within the study area presents the possibility that the groundwater may be overly exposed to contamination from mining waste, hence more vulnerable to contamination. The analysis of the groundwater flow direction using linear interpolation based on groundwater level data indicated that the groundwater flows towards a north-westerly direction. This groundwater movement confirmed to the flow of water from the surrounding mine dumps towards the boreholes at the study site. The study findings on fracture zones detection and groundwater flow direction were critical in contaminant tracing as well as in the development of the groundwater vulnerability model. The study results indicated that there was a high probability that all five boreholes in the study area emanated from the same shallow unconfined aquifer. The correlation between the direction of ground flow movement and the water quality was not very strong thus suggesting that other mine dumps might also have been responsible for contributing to the contamination of groundwater in the study area. Overall, while the north-westerly flow direction decreases the immediate risk of contamination from the nearby mine dump, further research into alternative contamination sources and the potential for localized flow direction changes is essential to fully understand and mitigate groundwater contamination risks in the study area.

Chapter 5

Proposed Groundwater Specific Vulnerability Concept for Abandoned Mine Dumps

5.1 Introduction

Recent research on groundwater vulnerability underscores the pressing requirement for establishing a comprehensive and adaptable methodology to assess specific groundwater vulnerability for source protection amidst fluctuating hydrogeological and hydro-climatic conditions. The application of several conventional groundwater vulnerability assessment methods in investigating the vulnerability of groundwater to contamination does not always bring desirable and accurate outcomes as different geological environments have unique characteristics that may not conform to some of the parameters used in the established vulnerability assessment techniques. It is thus imperative that a groundwater vulnerability assessment method applicable to the fractured unconfined aquifer within a mine dump be developed. The developed groundwater vulnerability assessment method will be specifically targeting how contaminants travel from the mine dump (source) to the groundwater (pathways) to the boreholes (receptors/targets).

5.2 Point source pathway receptor model

The availability of hydrological input data is one of the most important considerations when developing a groundwater vulnerability assessment method (Oke, 2015). The parameters for the input data in this study were developed to suit the conditions from the study area. The intention of developing the vulnerability assessment method is to gauge the risk posed by mine dumps in contaminating the groundwater sources located along unconfined fractured aquifers.

The proposed vulnerability model for this study relied on the source–pathway–receptor model that was initially developed by the European groundwater vulnerability approach, origin–pathway–receptor (Daly, 2002; Zwahlen, 2004). The source-pathway-receptor model has demonstrated its effectiveness in elucidating the movement of environmental contaminants from their origin, traversing various pathways to reach potential receptors.

The model has been applied to trace the movement of pesticides and fertilizers from agricultural fields, through soil and water pathways, to receptors such as rivers, lakes, and groundwater sources, helping in pollution control and mitigation strategies (Palladino et al., 2016; Zhang et al., 2017). Moreover, it has been widely utilized in environmental risk assessments concerning hazardous substances (Driscoll et al., 2013). The model developed in this study attempted to close the gap in assessing the groundwater vulnerability to contamination resulting from mine dumps in unconfined fractured aquifers by assessing the movement and transport of pollutants from the mine dumps identified as the source to the borehole water in the target area referred to as the receptor. Figure 5.1 shows the proposed Source–Pathway–Receptor concept.

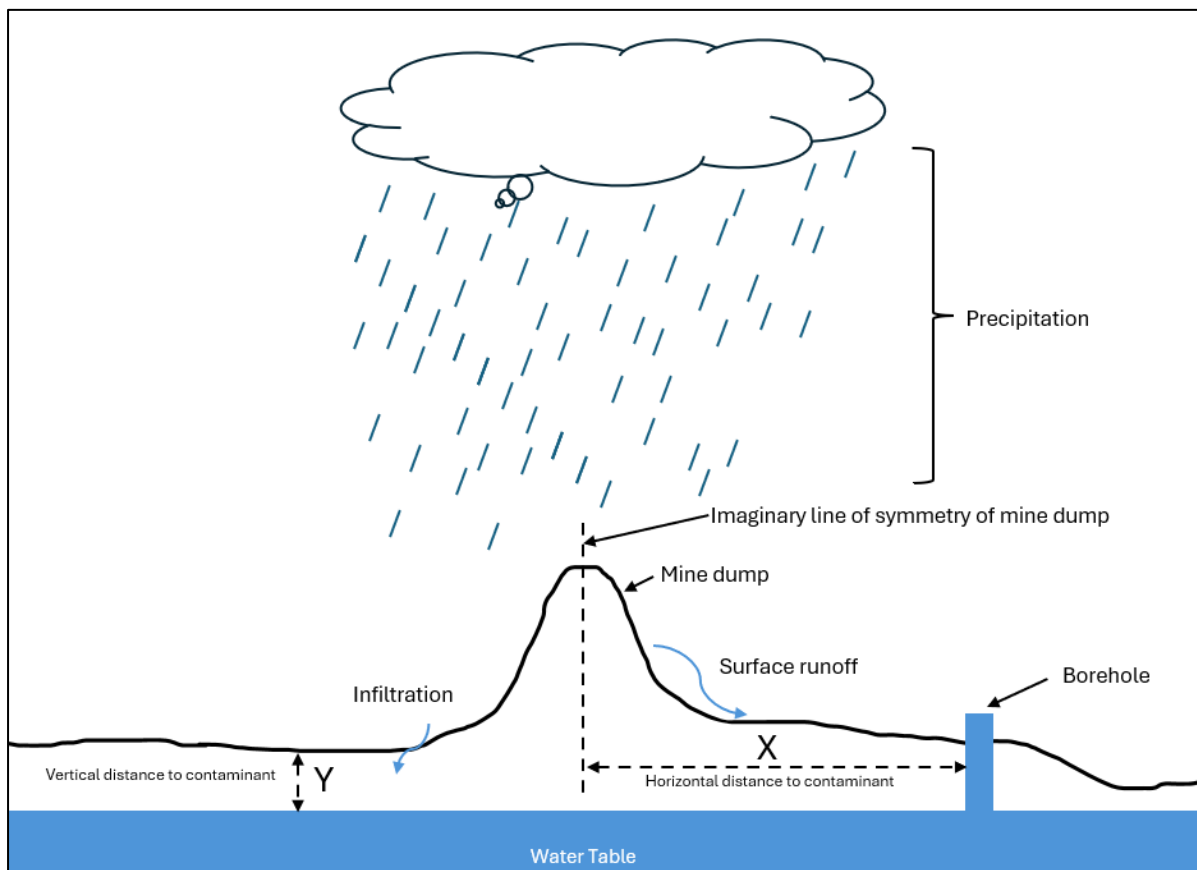
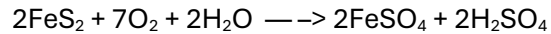


Figure 5.1 Adapted source–pathway–receptor concept approach (Author, 2023)

The concept assumes that precipitation through rainwater is the main driving force for the movement of contaminants from the mine dump into the underlying groundwater. Assuming that the rainwater falls on a mine dump from a gold mine or former gold mine, the rainwater infiltrates and percolates into the mine dump residue, where it reacts with pyrite and oxygen to form sulphuric acid as shown in Equation 5.1.



Equation 5.1

Where:

FeS_2 = Pyrite

O_2 = Oxygen

H_2O = Water

H_2SO_4 = Sulphuric acid

The sulphuric acid then dissolves, frees and washes away some of the metals, including heavy metals within the mine dumps. The resultant product of the reaction travels a vertical distance Y between the base of the mine dump residue and the water table, and a horizontal distance X before reaching the boreholes where it can be drawn out and utilised.

The proposed groundwater vulnerability model is based on the assumptions that the following five key parameters are critical in assessing the vulnerability of groundwater to contamination:

- **Precipitation:** Provides the force required to drive the pollutants from the mine dump.
- **Aquifer rock type:** Determines the nature, fate, and rate of infiltration of the surface water into the groundwater.
- **Horizontal distance between contaminant source and receptor (well or borehole):** Essential for determining the time as well as fate of pollutant travel.
- **Direction of flow of contaminants:** Indicates the possibility and probability of contamination of the receptors from the source.
- **Groundwater quality at the receptor(well or borehole):** Determines the extent of groundwater contamination.

The model can only be applied to shallow unconfined and or fractured aquifers and assumes that the vertical distance that a contaminant travels from the surface to the groundwater (Y) is catered for in the soil and rock structure underlying the aquifer, and is therefore less significant compared to the horizontal distance that a contaminant travels from the mine dump to the target borehole or well. The model also further assumes that:

- the abandoned mine dump is the source of point pollution;
- the mine dump and groundwater are hydraulically linked;

- there is negligible flow boundaries;
- the aquifer is fractured and or unconfined; and
- all the parameters have equal rating and weighting.

The characteristics of each of the five parameters of the proposed source–pathway–receptor model are explained in more detail in the following sections.

5.2.1 Water quality factor

Water quality is a general term that is used to describe the biological, chemical, physical, and thermal properties of water (Chakraborty and Kumar, 2016; Kroeze et al., 2016; Paun et al., 2016; Whitehead et al., 2009). In most cases, water quality is defined in terms of human consumption, aesthetics, and recreation. Shah et al. (2021) classified water quality into four types, namely potable water, palatable water, contaminated water, and infected water. Table 5.1 shows the different water quality types of classifications.

Table 5.1 Classification of water types

Type of water	Definition
Potable water	It is safe to drink, pleasant to taste, and usable for domestic purposes.
Palatable water	It is aesthetically pleasing; it considers the presence of chemicals that do not cause a threat to human health.
Contaminated water	It is water that contains unwanted physical, chemical, biological, or radiological substances, and it is unfit for drinking or domestic use.
Infected water	It is contaminated with pathogenic organisms.

Source: Shah et al. (2021)

This study focused on establishing the specific groundwater vulnerability assessment method for contaminated water with the main emphasis on the chemical and physical properties of the water in line with the SANS 241 drinking water quality guidelines.

The assessment of water quality is a critical component of evaluating the groundwater vulnerability to contamination. Gold and coal mining operations have been shown to adversely affect the groundwater and surface water quality through AMD (Jhariya et al., 2016; Linggasari et al., 2020; McCarthy, 2011). The proposed groundwater vulnerability assessment method thus focused on evaluating how these processes affect water quality.

In evaluating water quality, the proposed groundwater vulnerability model strongly relied on the assessment and prediction of the movement and transport of chemical elements and compounds released as by-products from the mining process. Mine dumps have been found to contain a unique chemical composition of metals, SO_4 , and heavy metals resulting as by-products from the mining and purification processes.

It is thus necessary for the groundwater vulnerability assessment methods to embody water quality parameters that can be measured against standard limits for drinking water quality guidelines such as SANS 241 and WHO (2022). In most cases, mine dumps are formed from the residue of crushed rocks after the extraction of minerals. In the case of gold, the crushed rock contains approximately 3% pyrite (iron disulphide) (Abdalla et al., 2010; Fashola et al., 2016). As rainwater falls onto the mine dumps, the pyrite is oxidised, thereby forming sulphuric acid. The sulphuric acid percolates into the mine dump, thereby dissolving and releasing large volumes of metals and heavy metals such as Cu, Ni, Zn, Hg, Cd, As, Pb, and Ur into groundwater and streams (Abengunde, 2014; McCarthy, 2011; Sakala et al., 2016). The polluted acidic water further infiltrates and contaminates the groundwater together with the linked aquifers. Apart from being acidic, the polluted water also has high concentrations of SO_4 (McCarthy, 2011).

For the water quality parameter to be included as part of the new specific groundwater vulnerability assessment method, several assumptions must be made:

- The infiltration and percolation of rainwater into the mine dump releases Fe and SO_4 , as well as heavy metals into the underlying groundwater aquifers. The concentration of Fe, SO_4 , and heavy metals in the borehole water will thus reflect on the extent of groundwater contamination.
- The water quality index formulated from Fe, SO_4 and the heavy metal concentrations identified in groundwater can be used as a proxy in groundwater vulnerability assessments.
- The fractured rock property types are permeable to contaminant movement and transport with very minimal adsorption.

As indicated in Chapter 4, the WAI method was chosen over other WQI calculation methods in the development of the model. This is mainly because WAI utilises less complicated calculations compared to the other WQI methods. Additionally, the WAI

method can be applied for single or multiple use measurements as the data are collected over a period of time.

5.2.1.1 Deriving water parameters for groundwater vulnerability assessment

As indicated earlier, the AMD process releases chemical compounds, metals, and heavy metal elements such as Fe, Cu, Ni, Zn, Hg, Cd, As, Pb, SO₄ and Ur into groundwater and streams (Abegunde, 2014; Sakala et al., 2016). It is thus important to utilise the WQI, which incorporates the physical and chemical parameters into the assessment of the groundwater vulnerability to contamination. The chemical parameters selected for use in the proposed groundwater vulnerability assessment method were pH, EC, Fe, NO₃, and SO₄. The choice of selecting these parameters were based on the following reasons:

- The physical and chemical variables can be easily measured and interpreted compared to biological parameters.
- The health risk factors of certain parameters as stipulated in the SANS 241 and WHO (2022) water quality guidelines.
- The high probability of finding the parameters in mine dumps, namely the presence of Fe, and SO₄ as byproducts of the gold processing process.

Table 5.2 shows the SANS 241 and WHO standard limits for drinking water quality guidelines for the proposed water quality parameters.

Table 5.2 Water quality parameters for the proposed vulnerability assessment against the SANS 241 (2015) and WHO (2022) water quality standards

Parameter	Limit standard (SANS 241)	Limit standard (WHO, 2022)	Risk
Electrical conductivity (EC)	170 mS/m	400 µS/cm	Acute health
pH	5–9.7	8.5	Operational
Nitrate (NO ₃)	11 mg/ℓ	50 mg/ℓ	Acute health
Sulphate (SO ₄)	500 mg/ℓ	250 mg/ℓ	acute health
Iron (Fe)	2 000 µg/ℓ	0.3 mg/ℓ	Chronic
	300 µg/ℓ		Aesthetic

5.2.1.2 Assigning a rating scale to the water quality index

For the water quality parameter to fit well into the proposed groundwater vulnerability assessment method, individual rating scales were assigned to each of the water quality index classes. The excellent scale with the WQI ranging from 0 to 25 was assigned a level 1-rating, while the WQI above 100 were assigned a rating of 5. Table 5.3 shows the rating scales adapted to the WAI method.

Table 5.3 Water class parameters for the Weighted Arithmetic Index method

Water Quality Index	Water Quality Class	Rating
0–25	Excellent	1
26–50	Good	2
51–75	Poor	3
76–100	Very poor	4
Above 100	Not suitable for drinking purposes	5

5.2.1.3 Assigning the weighting scale

Water quality is one of the most important characteristics for evaluating contaminated water (Paun et al., 2016; Whitehead et al., 2009). Among the four variables – precipitation, horizontal distance between contaminant source and receptor, direction of flow of contaminant, and the groundwater quality at the target – water quality assessment is critical in reflecting the extent of contamination. However, the proposed groundwater vulnerability assessment method assumes that all five parameters are equally important in contributing to groundwater vulnerability and no weighting values are therefore assigned to any parameter.

5.2.2 Precipitation factor

Precipitation in the Welkom area is usually in the form of rainfall. According to Oke (2015), the amount of rainfall received in an area is proportional to the rate at which contaminants migrate from the source (mine dump) to the groundwater under certain conditions. Although some mine effluent may be responsible for transporting contaminants from the mine dump to the groundwater, the proposed model considers rainfall as the primary mode of contaminant transport from the mine dump to the groundwater. This is mainly because rainfall:

- is mostly responsible for transporting contaminants on both surface water and groundwater in many natural processes; and
- data are readily available and easy to collect in many areas.

While many studies indicate that rainfall is responsible for transporting contaminants from surface water to groundwater, it is important to consider the effect that rainfall duration, intensity, and frequency has on contaminant transport and thus the groundwater vulnerability to contamination. Studies have demonstrated that the role of rainfall in transporting contaminants varied greatly between regions with extreme rainfall events such as flooding being linked to more widespread mobilisation of contaminants, and focused recharge of contaminated water into the groundwater, while it leads to more surface run-off and little contamination in other areas (Gowrisankar et al., 2017).

Oke (2015) emphasized that increased rainfall intensity typically leads to heightened surface flow, faster transport speeds, shorter transit durations, greater turbulence, enhanced sediment, and contaminant transport, including microbial pathogens and particle-bound chemicals, resulting in elevated susceptibility to groundwater contamination. Nevertheless, evidence suggests that even low-intensity rainfall events may contribute to widespread mobilization of contaminants, especially adsorbed contaminants in deep aquifers. Such aquifers are typically characterized by less frequent or slower groundwater renewal due to concentrated recharge and preferential flow and transport mechanisms (Hartmann et al., 2020).

For the purposes of this study, only the annual rainfall quantities will be considered over the Welkom area. This is because the Welkom area is considered semi-arid and receives low intensity rain with an average quantity of 577 mm per annum. The proposed method assumes that there will be very little to no surface run-off as both the intensity and quantity of the rain is low in the Welkom area.

5.2.2.1 Rating of the precipitation factor

The rating of the precipitation factor was primarily based on the annual quantity of rainfall received in the Welkom area. The amount of rainfall quantity was used as a proxy for the net recharge representing the volume of water reaching the groundwater aquifers. As highlighted by Barbulescu (2020), the possibility of groundwater contamination

increases with the net recharge. The three most common types of recharges are the direct, indirect, and localised types. The proposed model assumes that:

- the groundwater recharge rate is solely affected and hence directly proportional to the precipitation rate; and
- all three types of recharges operate at a constant and uniform rate in the contamination of groundwater in the Welkom area.

Rainfall categories were rated as shown in Table 5.4. The average annual rainfall exceeding 500 mm were rated with a scale of five, as their expected contribution to groundwater vulnerability to contamination will be high. Similarly, the scale of 200 mm was given the lowest rating as its expected to have a lower contribution to the vulnerability of groundwater to contamination.

Table 5.4 Rainfall rating of the proposed new vulnerability method

Rainfall (mm)	Rating
> 500	5
400–500	4
300–400	3
200–300	2
< 200	1

5.2.2.2 Assumptions of the relationship between recharge and rainfall

One of the major assumptions made in the development of the rating factor was the equating of the rainfall to recharge factors. This is because several studies have established a close relationship between the two variables. In their studies on establishing the relationship between rainfall and groundwater recharge in Cococodji, Benin, Kotchoni et al. (2019) observed a consistent seasonal and linear relationship between rainfall and groundwater recharge for a period of more than five years. Similarly, in their study on the effects of rainfall intensity on groundwater recharge based on simulated rainfall experiments and a groundwater flow along the Weihe River in China, Wang et al. (2015) concluded that rainfall had a great influence on recharge and groundwater regime flows.

Hussain et al. (2022) studied the relationship between 16 rainfall stations and 13 groundwater level stations to investigate the relationship between the groundwater level

response to rainfall and the recharge potential for shallow aquifers in the sub-regional area of Kaohsiung city in Taiwan. The results of the study indicated a positive linear correlation between groundwater level responses to associated rainfall.

Figure 5.2 shows the relationship between recharge and rainfall in Cocododji, Benin, as highlighted by Kotchoni et al. (2019).

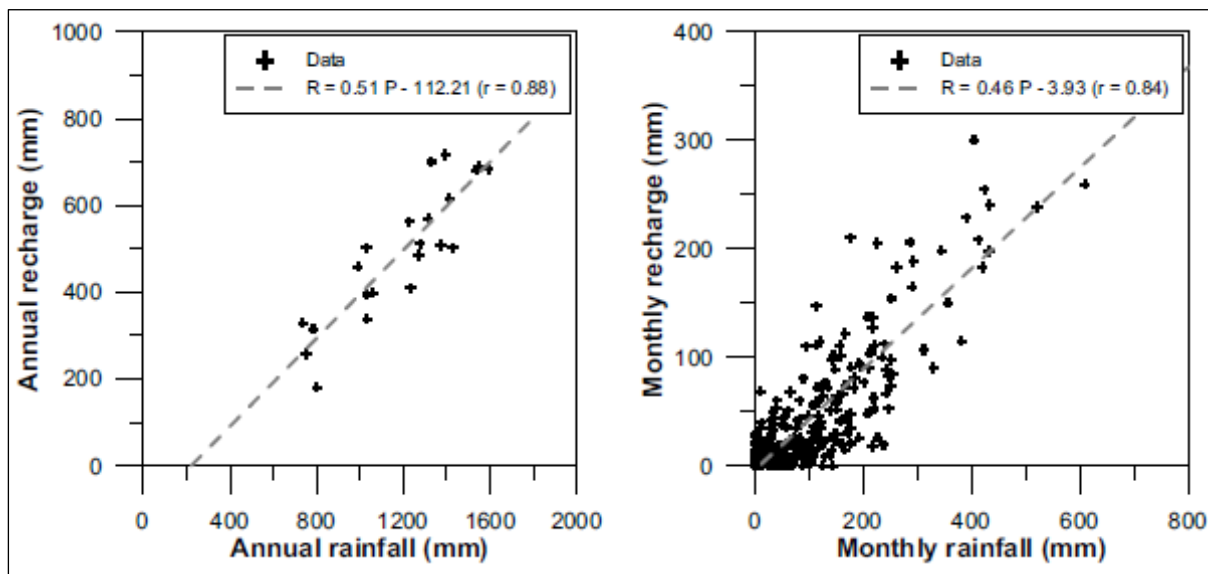


Figure 5.2 Relationships between recharge and rainfall in Cocododji, Benin (Kotchoni et al., 2019)

Studies conducted in Limpopo, South Africa, identified the estimated local recharge rates ranging from 0.24 to 8.78 mm/a using the Chloride Mass Balance method (Nemaxwi et al. 2019).

Despite several study findings indicating the positive correlations between recharge and rainfall, the relationship between rainfall and recharge is sometimes not always positive due to the extent of the rainfall intensity. Studies conducted by Wang et al. (2015) at the Weihe River, located between the arid and semi-arid regions of China, to assess the impact of rainfall intensity on groundwater regime, indicated a negative recharge on higher rainfall intensities. However, the rainfall intensities in most arid to semi-arid regions such as Welkom in South Africa is very low to moderate and therefore unlikely to strongly influence the correlation between the recharge and rainfall.

5.2.3 Horizontal distance between contaminant sources and abstraction points (targets)

Many studies investigating the specific vulnerability to contamination tend to overlook the significance of the horizontal distance between the contaminant source and the abstraction points in determining the extent of pollution from point sources. According to DWAF (2004b), groundwater contamination can occur through aquifers, and or preferential flow pathways. In the aquifer pathway, contaminants flow in the vertical distance before they reach the water table, and once they reach the water table, the contaminants travel a horizontal distance towards the abstraction or groundwater supply points.

Studies establishing the relationship between distance and the fate of chemical and microbial organisms have in most cases shown a perfect inverse correlation reflecting an increase in the number of microorganisms with a decrease in the distance to the contaminant source. An assessment on the impact of distance on the microorganism numbers conducted by Megha et al. (2015) in Kodyathur, Kerala, India, showed an increase in total coliform counts and faecal coliform counts with a decrease in distance between the wells and pit latrines. Similarly, Adetunji and Odetokun (2011) found an increase in the number of total coliform counts and total aerobic bacteria counts with decreasing distance to septic tanks and pit latrines in the Agbowo community, Ibadan, Nigeria.

The proposed groundwater vulnerability assessment method therefore assumes the importance of the horizontal distance travelled by contaminants over the vertical distance. This is because of the reasons that not all vertical distance contaminant flows result in the direct contamination of the groundwater supply at the abstraction points. The proposed groundwater vulnerability technique will be applied to shallow unconfined groundwater aquifers; hence, the method assumes that the vertical flow, although important, may not be significant in solving the groundwater vulnerability assessment challenges. In some cases, the preferential flow pathways created by excessive digging in mining areas leaves very little protection of the groundwater, as the soil layers that protect groundwater are rendered insignificant (Abdullah, 2020; Jhariya et al., 2016).

The proposed model assumes that the proximity of a contaminant to a water source can be used as an indication of assessing the extent of possible pollution and hence the groundwater vulnerability to contamination.

5.2.3.1 Set back distance and travel time concepts

Set back distances are defined as the hypothetical safe area zones between a possible contaminant source and an abstraction point such as a well or a borehole (Adetunji and Odetokun, 2011; Akaahan et al., 2010; DWAF, 2004a; Isikwue et al., 2011). Setback distances relate to the separation of distances in different geological categories to the survival time for pathogens as they travel through the soil and underlying aquifers. Most municipalities around the world use setback distances in creating guidelines and for assessing the specific groundwater vulnerability due to pit latrines, animal kraals, agriculture, burial sites, and dump sites. The underlying assumption on the use of setback distance is that the travel distance of a microbial or chemical contaminant can be treated as a renewal process (Haith and Laden, 1989). The proposed groundwater vulnerability assessment method has adapted the use of setback distances by developing a distance parameter through which a set of different ranges of distances are assigned numerical ratings in relation to their proximity to the contaminant source. Figure 5.3 shows an example of a setback chart with protection zones from a possible contaminant source.

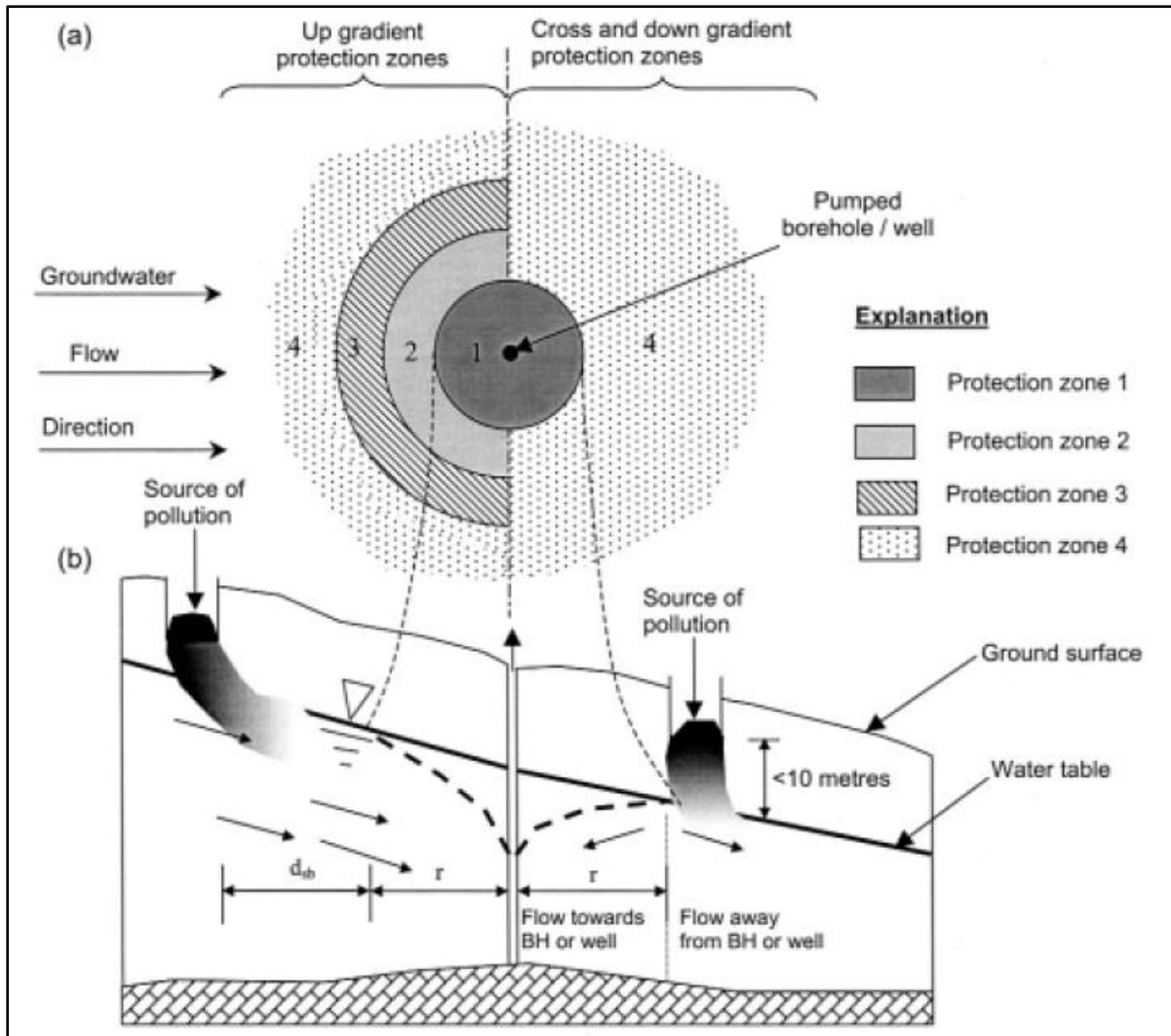


Figure 5.3 Example of a setback chart with protection zones from a possible contaminant source (DWAf, 2004b)

5.2.3.2 Adaptation of the distance concept to groundwater vulnerability assessment in mine dumps

Despite the setback concept being mainly applied by local governments to regulate and minimise the contamination of groundwater by specific pollution point sources such as pit latrines, and animal kraals, the proposed groundwater vulnerability concept will utilise the same principle in assessing the vulnerability of groundwater to contamination from mine dumps (Still and Nash, 2005). The proposed groundwater vulnerability concept will adapt and use the horizontal distance parameter based on the setback safety guidelines distances. The principles that influence the development of groundwater to distance parameters are:

- The survival times for pathogens.

- The ranges of survival times for pathogens.
- The range of travel times for physical and chemical elements.

Table 5.5 shows a summary of the bacterial transport through soils. The summary was developed by Crane and Moore (1984) and Still and Nash (2005) and is used as a guide for informing the residence and travel times of different microorganisms under specific point pollution sources.

Table 5.5 Summary of studies of bacterial transport through soils

Nature of pollution	Organisms	Medium	Measured distance travels	Time of travel	Reference
Sewage trenches intersecting groundwater	<i>Bacillus coli</i>	Fine sand	19.8 m	27 wks	Stiles and Crohurst (1923)
Sewage trenches intersecting groundwater	Coliforms	-	70.7 m	-	Warrick and Muegge (1930)
Sewage in pit latrine intersecting groundwater	<i>Bacillus coli</i>	Fine and coarse sand	24.4 m	-	Caldwell (1937)
Sewage in pit latrine intersecting groundwater	<i>Bacillus coli</i>	Sand and sandy clay	10.7 m	8 wks	Caldwell and Parr (1937)
Sewage in pit latrine intersecting groundwater	<i>Bacillus coli</i>	Fine and medium sand	3.1 m	-	Caldwell (1938)
Primary and treated sewage in infiltration basins	Coliforms	Fine sandy loam	0.6-4 m	-	Butler et al. (1954)
Diluted primary sewage injected subsurface	Coliforms	Aquifer	30 m	33 hrs	McGauhey and Krone (1954)
Canal water in infiltration basins	<i>Escherichia coli</i>	Sand dunes	3.1 m	-	Baars (1957)
Subsurface injections	Enterococci	-	15 m	-	Fournelle (1957)
Secondary sewage in infiltration basins	Coliforms	Sandy gravels	0.9 m	-	McMichael and McKee (1966)
Tertiary treated wastewater in percolation beds	Faecal coliforms and faecal streptococci	Coarse gravels	457.2 m	15 da	Merrell (1967)
Primary sewage injected subsurface	Coliforms	Sand and pea gravel aquifer	30.5 m	35 hrs	Krone et al. (1958)
Secondary sewage injected subsurface	Faecal coliforms	Fine to coarse sand aquifer	30.5 m	-	Wesner and Baier (1970)
Tertiary treated wastewater in percolation beds	Coliforms	Sand and gravel	20 m	-	Ana'ev and Demin (1971)
Inoculated water and diluted sewage injected subsurface	<i>Bacillus stearothermophilis</i>	Crystalline bedrock	13.5 m	24-30 hr	Allen and Morrison (1973)
Tertiary treated wastewater in infiltration basins	Coliforms	Fine to medium sand	6.1 m	-	Young (1974)
Secondary sewage in infiltration basins	Faecal coliforms	Fine loamy sand to gravel	9.1 m	-	Bower et al. (1974)
Primary sewage in infiltration basins	Faecal streptococci	Silty sand and gravel	183 m	-	Schaub and Sorber (1977)
Secondary sewage effluent in infiltration basins	Faecal coliforms and faecal streptococci	Fine loamy sand to gravel	9 m	-	Gilbert et al. (1976)
Septic tank tile effluent	Total coliforms Faecal coliforms	Fine loamy soil	6.1 m 13.5 m	-	Reneau and Pettry (1975)
Inoculated effluent in tile line	<i>E. coli</i>	Silty clay loam	20 m	5 hr	McCoy and Hagedorn (1979)

Source: Crane and Moore (1984); Still and Nash (2005)

The data on the residence and travel times of microorganisms and physicochemical elements were used to guide the processes of developing distance to contamination charts for the new groundwater vulnerability assessment method. Studies on groundwater vulnerability from point source pollution sources have indicated an almost perfect negative correlation between distance and concentration of physicochemical parameters found in groundwater. Studies conducted by Oyiboka (2014) on the effects of distance on heavy metal distribution in a landfill site in Igando, Lagos, Nigeria, found a

negative correlation between many metals such as Pb, Zn, and Mn and distance as shown in Table 5.6.

Table 5.6 Relationship between metals and distance

	Variable	Fe	Mg	Cu	Zn	Pb	Cr	Mn	Distance
1	Fe	1							
2	Mg	0.298	1						
3	Cu	0.21	0.193	1					
4	Zn	0.002	-0.117	-0.073	1				
5	Pb	0.018	0.096	-0.101	0.998*	1			
6	Cr	0.986*	0.21	0.27	-0.059	-0.048	1		
7	Mn	-0.061	-0.147	-0.402	-0.078	-0.035	-0.078	1	
8	Distance	0.057	0.021	0.927*	-0.016	-0.043	0.138	-0.350	1

*Correlation is significant at the 0.01 level (two-tailed).

Source: Oyiboka (2014)

Another study conducted by Kanda et al. (2021), in tracing the distance covered by trace elements from a mine dump in Bindura, Zimbabwe, also revealed a negative relationship between distance and elements such as sulphate ion (SO_4^{2-}), Cl, Cr, and Ni as shown in Table 5.7.

Table 5.7 Variation of water quality parameters with distance

Factor		SO_4^{2-}	Cl	Cd	Cu	Cl	Cr	Ni
Depth	<i>R</i>	-0.78	-0.63	-0.81		-0.85	-0.75	-0.78
	<i>R</i> ²	0.61	0.40	0.66		0.72	0.56	0.61
Distance	<i>R</i>	-0.83	-0.7		-0.79	-0.77	-0.79	-0.70
	<i>R</i> ²	0.69	0.49		0.62	0.59	0.6241	0.49

Source: Kanda et al. (2021)

The range of distance from the contaminant source was divided into five classes, each with a difference of approximately 1 000 m. The ratings were based on subjective observations from other studies as well as the distances travelled by physical, chemical, and microbiological organisms (Kanda et al., 2021; Oyiboka, 2014). Figure 5.4 shows the rating of the distance to contaminant from the source. Boreholes and wells in the R5 region were more vulnerable to contamination as compared to boreholes and wells in the R1 region due to their proximity to the point source pollution sources.

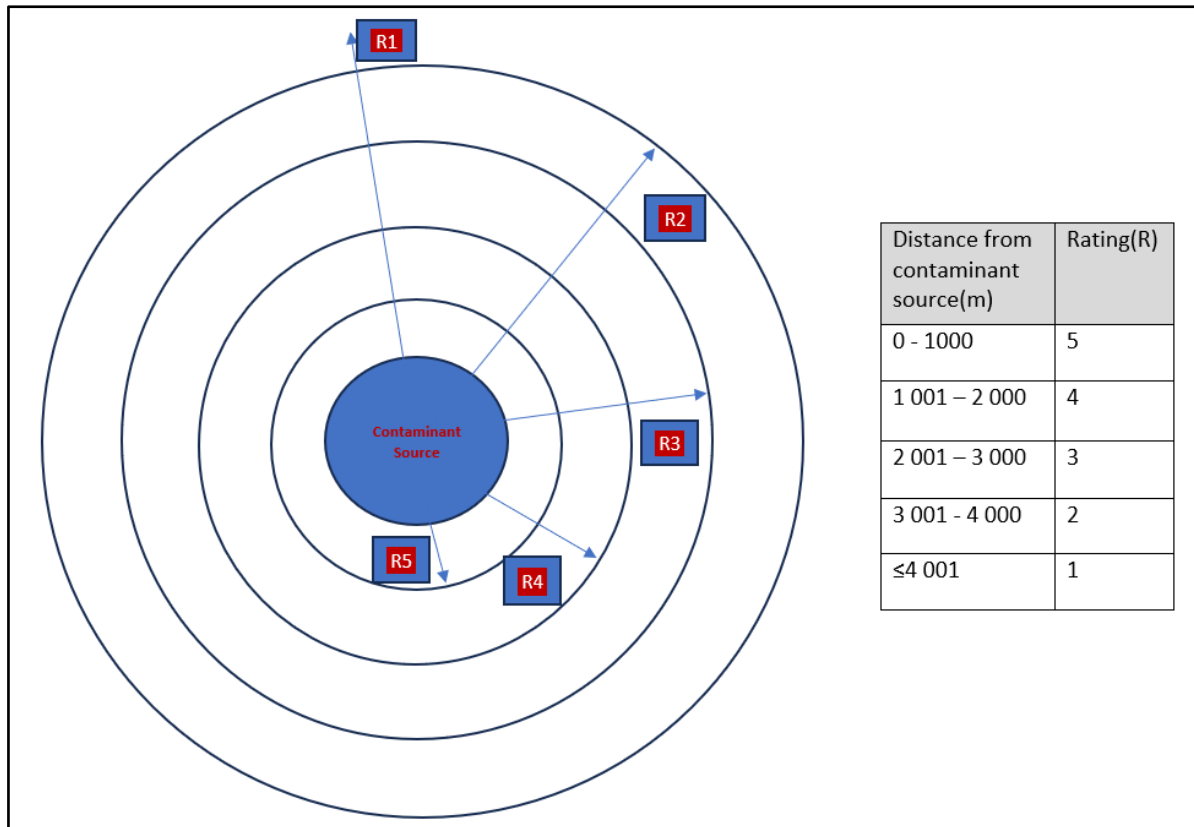


Figure 5.4 Rating of the distance from contaminant source (Author, 2023)

5.2.4 Direction of groundwater flow between contaminant source and target

One of the most important factors of the proposed groundwater vulnerability assessment method is its consideration of groundwater flow direction. The assumptions that justify the inclusion of this parameters are that:

- groundwater flow direction is critical in mapping the risk to pollution of target sources; and
- the accuracy of the groundwater flow direction in tracing specific point pollution supersedes that of topography and slope (Nimer, 2020).

Unlike surface water, groundwater flow direction cannot be predicted with ease as it cannot be seen. Groundwater flows from areas with a high hydraulic heads to areas with a low hydraulic head as it utilises hydraulic pressure differences (Kotchoni et al., 2019). Groundwater flow may therefore not be consistent with the surface flow direction that is guided by surface gradient or topography. The inclusion and use of the hydraulic head to determine groundwater flow direction therefore gives the proposed groundwater

vulnerability assessment technique an advantage over other techniques that only rely on surface gradient or topography, such as the DRASTIC method.

To determine the direction of groundwater flow relative to the contaminant source, the groundwater flow direction needs to be initially determined through a series of steps as indicated in Chapter 4, section 4.3.2.5. Once the groundwater flow direction is established, the direction of groundwater flow relative to the contaminant can be assessed. Regardless of the location of the contaminant, the direction of flow can be mapped up in a backward direction from the well.

In the proposed groundwater vulnerability assessment method, any groundwater flow away from the contaminant source to the well is deemed as negative flow, while any flow away from the well towards source directions are deemed as undefined groundwater flows. Figure 5.5 shows the characterisation of the groundwater flow direction between a well and a contaminant source.

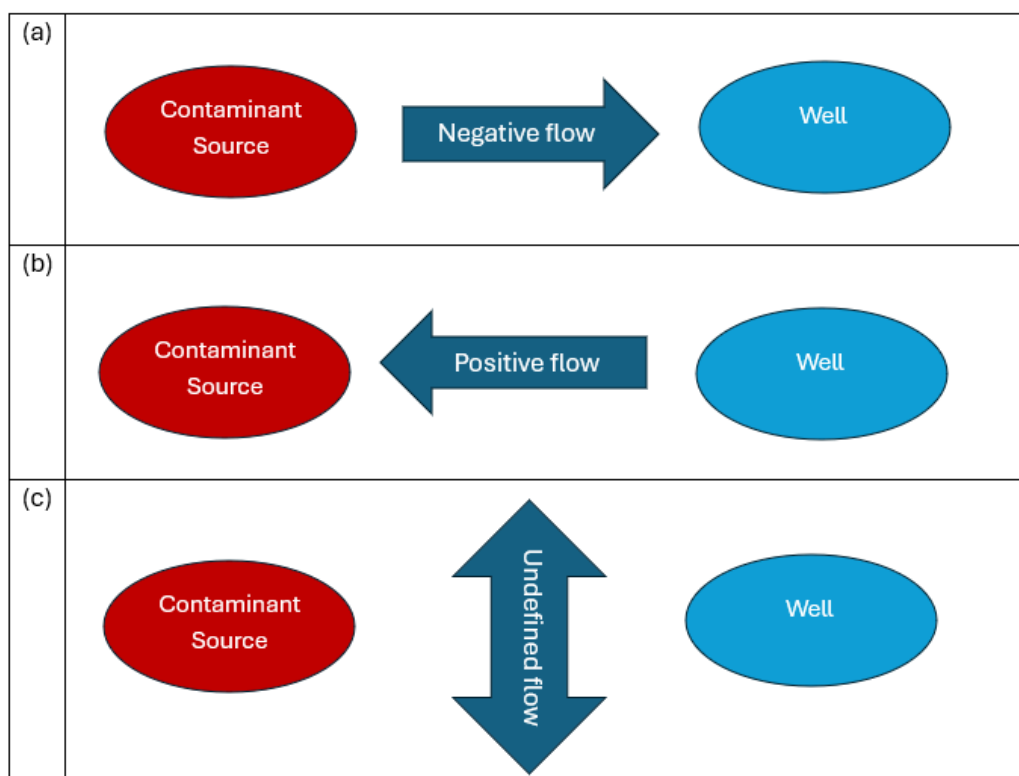


Figure 5.5 Characterisation of groundwater flow direction patterns between a well and contaminant source: (a) Negative flow, (b) Positive flow, (c) Undefined flow (Author, 2023)

Like many other parameters used in groundwater vulnerability assessment models, the groundwater flow direction parameter was designed to work under a set of assumptions. The assumptions on groundwater flow directions are:

- 1) **Negative flow:** The well lies in the contaminant flow paths and direction, such that there is greater probability or risk for contamination.
- 2) **Positive flow:** The direction of contaminant flow lies in an opposite direction to the position of the well, and the risk for contamination is very low.
- 3) **Undefined flow:** The direction of contaminant flow lies perpendicular to the position of the contaminant source and the well such, that the risk of contamination cannot be determined with certainty.

A consolidated view of the groundwater flow directions and characteristics in relation to the contaminant source is shown in Figure 5.6.

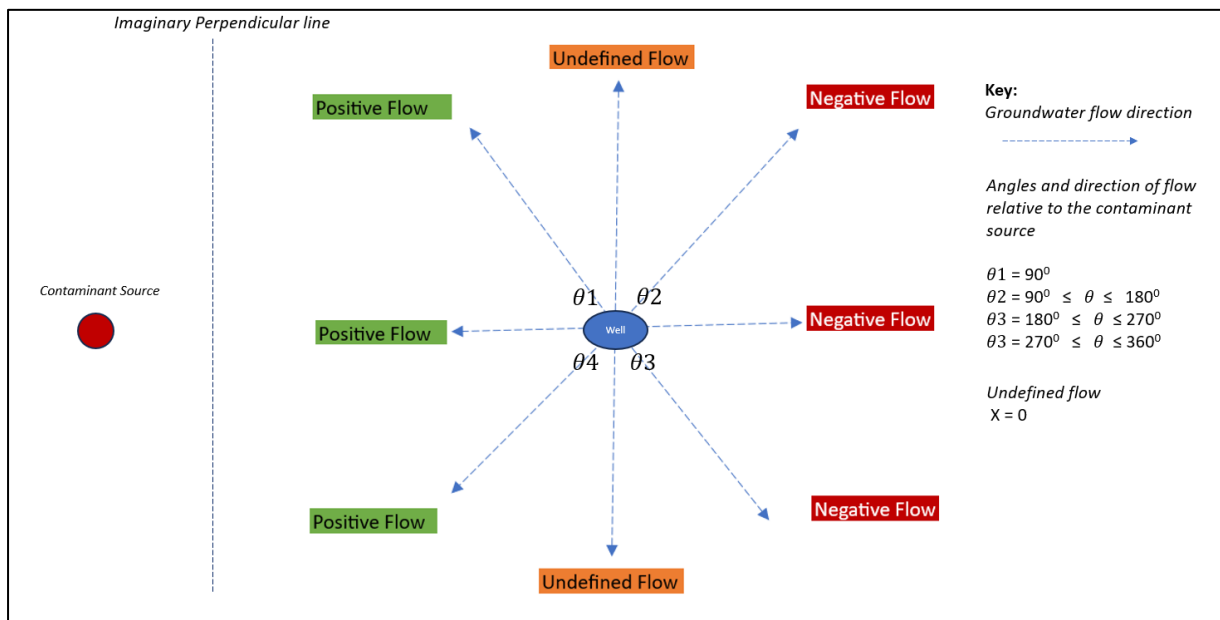


Figure 5.6 Consolidated chart on the characterisation of groundwater flow direction (Author, 2023)

5.2.4.1 Rating of groundwater flow direction

Based on the groundwater flow direction and the characterisation from the contaminant source, subjective ratings were assigned to the different groundwater flow directions. The groundwater flow characterised as positive flow was assigned a vulnerability class of 1. Groundwater flow in the undefined category was assigned a rating of 3, and groundwater flow direction in the negative flow direction was assigned a rating of 5. The subjective ratings were merely based on the assumptions of groundwater flow directions between the supposed contact source and the well or abstraction point. Table 5.8 shows the assigned ratings attached to the groundwater flow direction chart.

Table 5.8 Assigning the rating scale to groundwater flow direction

Classification of groundwater flow direction	Positive flow (Direction of flow is away for the well) $0^\circ \leq \theta \leq 90^\circ$ ($270^\circ \leq \theta \leq -360^\circ$)	Undefined flow (Direction of flow is undefined) $\theta = x = 0$	Negative flow (Direction of flow is towards the well) $90^\circ \leq \theta \leq 180^\circ$ $180^\circ \leq \theta \leq 270^\circ$ $Y = 0$
Vulnerability class	Low vulnerability	Medium vulnerability	High vulnerability
Rating	1	3	5

5.2.4.2 Limitations on the application of the groundwater flow direction factor

Groundwater flow direction is a reliable parameter other than slope or topography as it uses the real depth to groundwater measurements to estimate the possible groundwater flow direction between a well and a contaminant source. However, while the groundwater flow direction factor is a valued addition to the proposed groundwater vulnerability assessment method, its applications come with certain limitations. Some of these limitations are the following:

- The predictions on groundwater flow direction are not always reliable. Flow directions may change depending on the underlying soil and rock formations.
- The calculations involved in the process of determining groundwater flow directions may be challenging if not done well and may lead to inappropriate conclusions.

5.2.5 Aquifer rock type

The aquifer rock type plays a significant role in determining the groundwater vulnerability to contamination. An aquifer is normally defined as a body of saturated rock with the capacity to store water (Barbulescu, 2020; Shah et al., 2021). The vulnerability of an aquifer is strongly influenced by its type and the nature of the geology that surrounds it.

Nimer (2020) highlighted that aquifer rock type is important for groundwater recharge. In most instances, the aquifer medium controls the flowrate, which provides valuable insights on the contamination rate. Aquifers with larger grain sizes, high void ratios, and more fractures exhibit higher permeability rates, reducing their ability to attenuate

pollutants and potentially increasing their contamination potential (Mazdarani et al., 2023; Pouye et al., 2022).

The impact of rock types on groundwater vulnerability hinges on their hydraulic conductivity and permeability. Unfractured basement rock typically exhibits minimal vulnerability, whereas the vulnerability of fractured basement rock is contingent upon factors like fracture distribution, width range, and frequency. The vulnerability of fractured rock is directly influenced by its hydraulic conductivity, resulting in a slightly lower rating compared to fresh, unfractured rocks. Weathered basement rocks, characterized by increased pore spaces, carry a higher vulnerability weight. In contrast, consolidated sedimentary rocks like shale and claystone have reduced pore space and very low permeability.

In the proposed model, various rock types were assessed and adjusted according to a contextual rating scale ranging from 1 to 5. A rating of 1 was assigned to rock types with minimal infiltration capacity and higher surface runoff, such as consolidated dense rock, shale, igneous, and metamorphic rocks. Unconsolidated sedimentary rocks received a rating of 5 due to their ability to facilitate quicker water infiltration into underground water sources. These rocks typically exhibit the highest hydraulic conductivity among geological formations, attributed to their large pore spaces and greater permeability. Oke (2015) conducted a study on the Dahomey basin in Western Nigeria, rating the aquifer rock types as shown in Table 5.9 using the RTt model, with the modified rating table presented in Table 5.10.

Table 5.9 Aquifer rock types

Rock type	Rating
Consolidated dense rock, shale, igneous and metamorphic rocks	5
Fractured igneous and metamorphic	4
Sandstone, siltstone, and limestone	3
Weathered basement	2
Unconsolidated sediment	1

Source: Oke (2015).

Table 5.10 Adaptation of the aquifer rock types ratings

Rock type	Rating
Consolidated dense rock, shale, igneous and metamorphic rocks	1
Fractured igneous and metamorphic	2
Sandstone, siltstone, and limestone	3
Weathered basement	4
Unconsolidated sediment	5

Source: Oke (2015).

5.3 Development of the DWAPH method

The DWAPH method is a parametric model developed to assess the specific vulnerability of groundwater to contamination from a mine dump source. Its design is based on the source–pathway–receptor model. Machiwal et al. (2018) argued that more studies should be devoted to developing vulnerability assessment techniques for source protection through using the source-pathway-receptor model. The design of the DWAPH method is based on five parameters, which are assumed to be critical in influencing the vulnerability of groundwater to contamination from a point source. These parameters are:

- D – Direction of groundwater flow.
- W – Water quality.
- A – Aquifer type.
- P – Precipitation.
- H – Horizontal distance to contaminant source.

The DWAPH method is a parametric method that is designed to be applied and interpreted with ease by researchers at all levels. All five parameters were assigned the subjective ratings from 1 to 5, with Level 1 assumed to be contributing less risk to contamination (low vulnerability) of the groundwater and Level 5 contributing to more risk of contamination (high vulnerability) to groundwater.

The choice of selecting the five parameters for this method was made on the basis that

- each of the five parameters is easily accessible in data scarce areas;
- each of the parameters can be measured and calculated with ease and without the use of complicated instruments; and

- each of the parameters provides meaningful guidelines in identifying and estimating specific groundwater vulnerability to contamination.

Each of the five parameters of the proposed vulnerability technique plays a significant role that fits well within the source–pathway–target model. The inclusion of the water quality parameter increases the sensitivity of the overall test outcome, while the distance parameter gives an indication of the probability and time factors of the contaminant reaching the target from the source. The DWAPH method is applicable in areas where determining the water quality data is essential.

The modifications and inclusions of the distance, water quality, and direction of flow parameters thus came with addressing some of the weaknesses in most parametric methods, including the DRASTIC method. According to Oke (2015), some of these weaknesses included the following:

- The lack of clarity on assessing specific type pollutants from point sources.
- Redundancy of factors: Some of the hydraulic factors such as conductivity are redundant as they are dependent on the aquifer medium.

There is an over-emphasis on the use of *slopes*, which are also catered for under topography. The use of both slopes and topography was, however, improved through the use of hydraulic gradient calculated directly from the depth to groundwater values.

- The index scores utilised in the DRASTIC method do not cater for continuous data.

Each of the five DWAPH parameters together with their ratings are shown in Table 5.11.

Table 5.11 The DWAPH model factors with the classification and ratings

Number	Factor	Class	Rate
1	Direction of groundwater flow (D)	Negative flow direction	1
		Undefined flow direction	3
		Positive flow direction	5
2	Water quality (W)	Excellent	1
		Good	2
		Poor	3
		Very poor	4
		Not suitable for drinking purposes	5

Number	Factor	Class	Rate
3	Aquifer type (A)	Consolidated dense rock, shale, igneous and metamorphic rocks	1
		Fractured igneous and metamorphic	2
		Sandstone, siltstone, and limestone	3
		Weathered basement	4
		Unconsolidated sediment	5
4	Precipitation (P) (average mℓ)	>500	5
		400–500	4
		300–400	3
		200–300	2
		<200	1
5	Horizontal distance to contaminant (H)	0 – 1 000	5
		1 001 – 2 000	4
		2 001 – 3 000	3
		3 001 – 4 000	2
		≤4 001	1

5.3.1 Limitations of the DWAPH method

Like any other groundwater vulnerability assessment method, the DWAPH method is based on a set of assumptions and has got its own limitations. Some of the limitations to the DWAPH method are:

- It can only be applied to shallow unconfined aquifers.
- All the parameter ratings are subjective and may be interpreted differently.
- It cannot be applied in assessing general groundwater vulnerability to contamination.
- The model introduces new variables which are water quality; direction of groundwater flow; and horizontal distance to contaminant source, with the assumption that they have a significant contribution to specific groundwater vulnerability.
- All five parameters are assigned equal ratings, and no weights, as all the parameters are considered to have an equal contribution to groundwater vulnerability.

However, despite these limitations, the DWAPH vulnerability assessment method can still be valuably applied in assessing groundwater vulnerability.

5.3.2 Vulnerability scale mapping

The range of the scale was calculated using the minimum and maximum parameter values as follows:

$$\begin{aligned} DWAPH_{\min} &= D_r + W_r + A_r + P_r + H_r \\ &= 1 + 1 + 1 + 1 + 1 \\ &= \mathbf{5} \end{aligned}$$

$$\begin{aligned} DWAPH_{\max} &= D_r + W_r + A_r + P_r + H_r \\ &= 5 + 5 + 5 + 5 + 5 \\ &= \mathbf{25} \end{aligned}$$

Range of vulnerability scale (x) = $5 \leq x \leq 25$

Table 5.12 shows the proposed range of the groundwater vulnerability scale.

Table 5.12 Vulnerability scale for the DWAPH method

Vulnerability class	Low vulnerability	Moderate vulnerability	High vulnerability	Very high vulnerability
Range (x)	5–10	11–15	16–20	21–25

The scale is divided into four range categories starting from very low vulnerability to very high vulnerability. The scale is entirely subjective and based on the ratings of the five parameters.

The overall DWAPH vulnerability index is thus given by the following equation:

$$DWAPH_{index} = D_r + W_r + A_r + P_r + H_r \quad \text{Equation 5.1}$$

Where:

- D_r = Direction rating of groundwater flow
- W_r = Water quality rating
- A_r = Aquifer type rating
- P_r = Precipitation rating
- H_r = Horizontal distance to contaminant source rating

The proposed DWAPH vulnerability method will be tested in Chapter 6.

5.4 Conclusion

The DWAPH groundwater vulnerability assessment method offers a straightforward parametric approach for evaluating the specific vulnerability of groundwater to contamination, particularly from mine dumps in arid to semi-arid regions. It incorporates five key parameters: groundwater flow direction, water quality, aquifer type, precipitation, and horizontal distance to the contaminant source, to create a vulnerability index. The method emphasizes precipitation as the primary driver for transporting contaminants from mine dumps to groundwater, while groundwater depth is linked to flow direction. Like many vulnerability assessment methods, DWAPH follows the source–pathway–receptor model, with parameter ratings based on a combination of subjective site observations and past studies. The resulting vulnerability index is divided into four classes, ranging from very low to very high vulnerability. Because of its simplicity, the DWAPH method can be applied in areas with limited data and minimal resources without compromising on its effectiveness. Figure 5.7 shows a consolidated flow diagram of the DWAPH model.

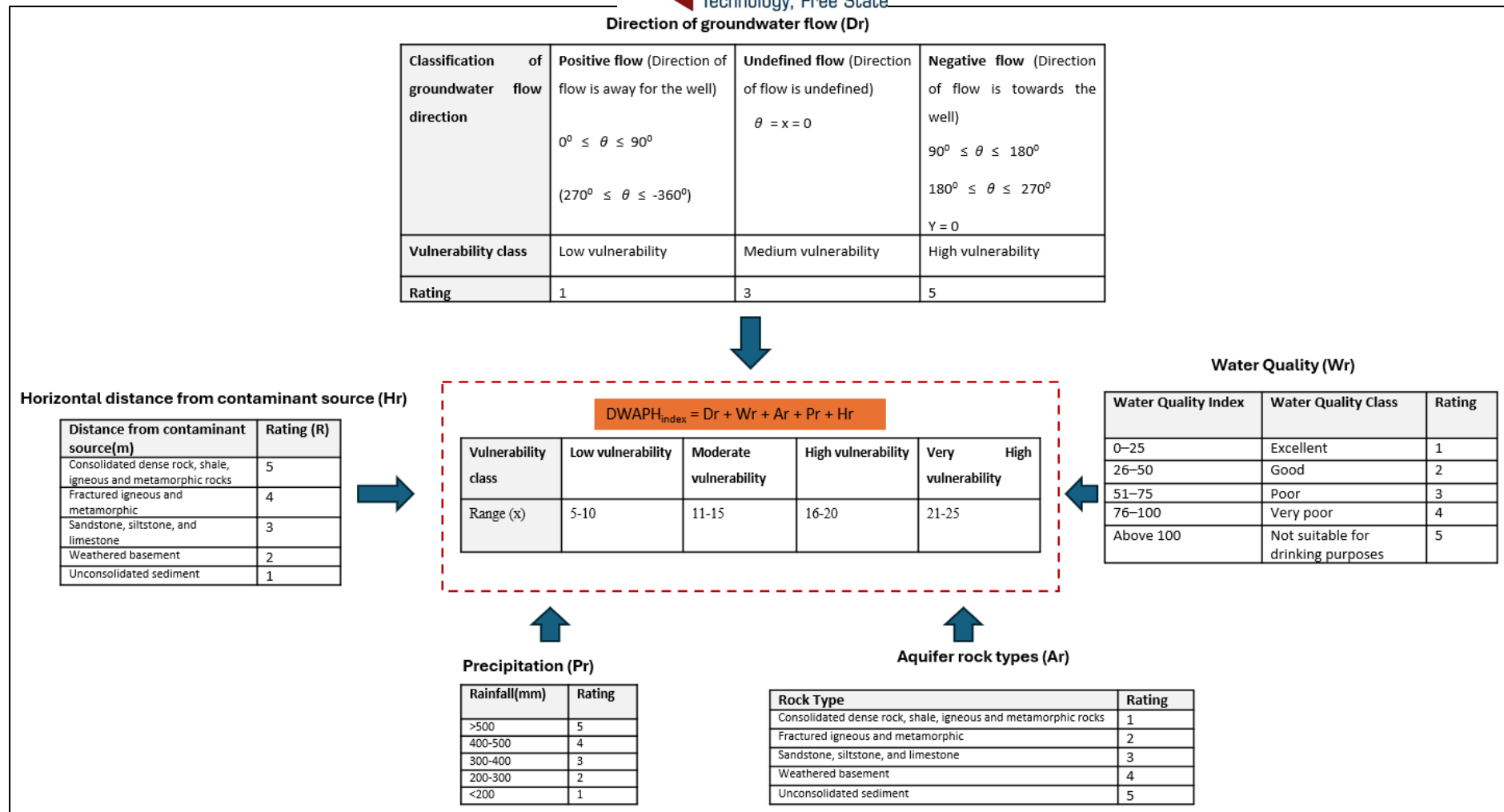


Figure 5.7 Flow diagram of the proposed DWAPH groundwater vulnerability assessment method

Chapter 6

Application of the DWAPH Model to Assess the Impact of Mine Dumps on Groundwater Vulnerability in the Welkom Area

6.1 Introduction

The DWAPH vulnerability method was applied to assess the specific vulnerability of groundwater to contamination from a mine dump in the Welkom area. As previously stated, the main concept of the DWAPH vulnerability assessment method relies on the source–pathway–receptor concept in which five parameters, including precipitation; water quality; aquifer type; horizontal distance travelled by the contaminant; and direction of contaminant movement are involved. Five boreholes located within the CUT Welkom campus were used as proxies in assessing the extent of contamination and to represent the larger Welkom area. Precipitation was assumed to be the main driving force for the movement of the contaminants from the mine dump to the boreholes.

The data for the five parameters were generated through cost-effective techniques from in-field measurements, laboratory analysis, and existing geological and topographical maps. Table 6.1 shows how each of the parameters measured were sourced for this study.

Table 6.1 Description of parameters and sources of data

Parameter	Description	Source
Direction of contaminant movement	The overall groundwater flow direction was calculated using depths to water of the five boreholes located at the CUT Welkom campus	Data was generated from in-field measurements of groundwater levels. The data was then interpolated to find the direction of groundwater flow
Water quality	The average water quality of the borehole water was calculated over a three-year period	Data generated from in-field measurements and analysis of water quality parameters.
Aquifer type	The classification of the underlying rock type of the Welkom aquifer	Data was generated from the Welkom geological maps provided by the South African Geological Council
Precipitation	The average annual precipitation of the Welkom area over a 30-year period	Data was obtained from the South African Weather Services (SAWS)
Horizontal distance to contaminant source	Average distance between boreholes at the CUT Welkom campus and the mine dump	Data generated from a topographical map, Google Maps, and field measurements

Out of the five parameters used in the DWAPH method, three parameters – direction of groundwater flow, water quality, and horizontal distance to contaminant flow – were used on-site for their measurements. The data for the nature of the aquifer type and precipitation, utilised historical information obtained from previous studies and data obtained from the SAWS.

6.1.1 Map construction and colour-coding

Each of the five parameters were plotted on a study map chart indicating the extent of *vulnerability* of point sources (boreholes) in relation to the parameter scales and available data. The study maps for each parameter were extracted from Google Maps using the following steps:

Step 1: Using the Google Maps function, the map of the study area was identified and extracted from the map of South Africa, the map of the Free State Province, and the map of Welkom. Figure 6.1 shows the extension of the maps across the four levels with the study area map having five boreholes and the mine dump.

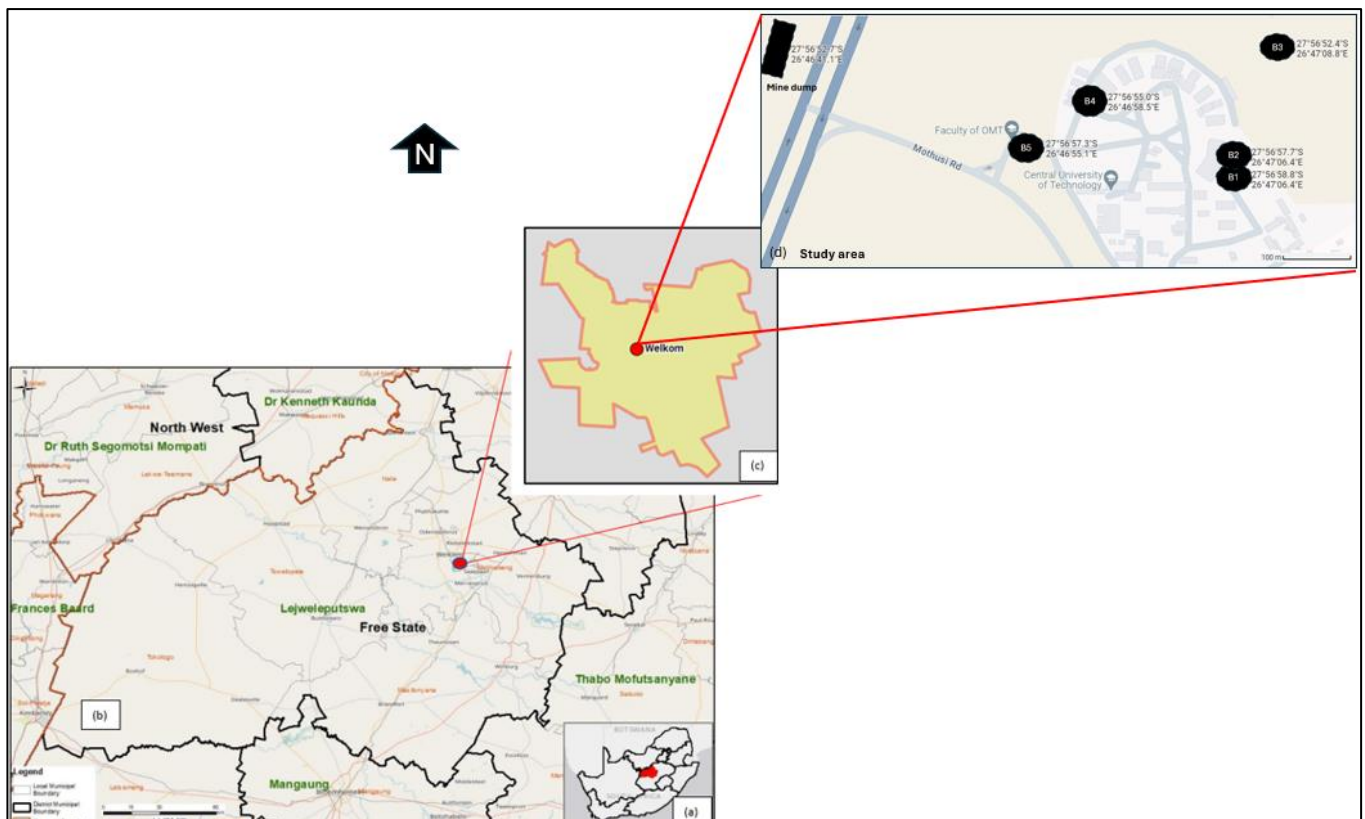


Figure 6.1 Map extraction from Google Maps showing (a) Map of South Africa, (b) map of the Free State Province, (c) map of Welkom, and (d) study area map of Welkom

Step 2: The location of the boreholes under investigation were identified and colour-coded depending on the outcome from the parameter for that location. The colour-coding utilised five colour criteria ranging from light green, dark green, blue, orange, and red. The ascendancy of the colour “depth” from green to red indicates an increasing risk of that parameter negatively affecting the quality of the groundwater. Figure 6.2 shows the alignment of colour codes and risk profiles.

Colour code	Risk
	Very High
	High
	Moderate
	Low
	Very Low

Figure 6.2 Alignment of colour codes and risk profiles

Step 3: The colour codes were merged with the parameter ratings, which were aligned to specific points within the study area maps. Figure 6.3 shows the association between the colour codes and rating scales.

Colour code	Risk	Rating
	Very High	5
	High	4
	Moderate	3
	Low	2
	Very Low	1

Figure 6.3 Alignment of colour codes, risk profiles, and ratings

Step 4: The ratings were applied on a map together with the study area data for each parameter. The groundwater data for each borehole parameter were analysed and assigned a rating and colour code as shown in Figure 6.4.

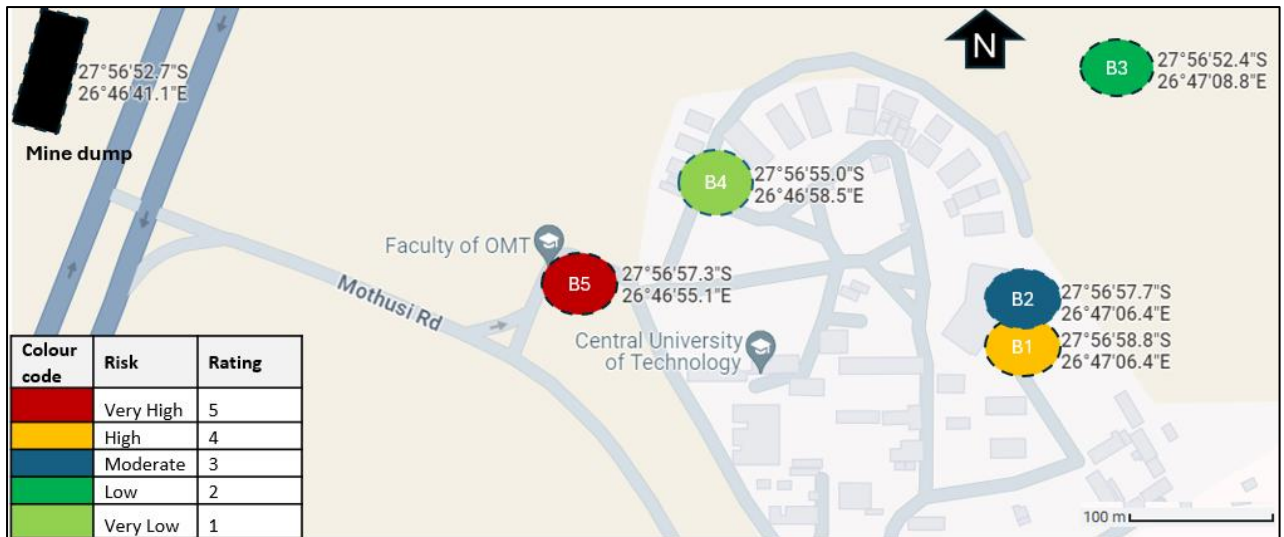


Figure 6.4 An example of a parameter risk map for the study area

Step 5: The study area was further analysed using vulnerability maps created through the Bing GIS software using Microsoft Excel. The map of the study area was secluded through the use of the geographical coordinates for each point. Figure 6.5 shows the study area map created from GIS. The same colour code used in the map development was used in the development of GIS maps.

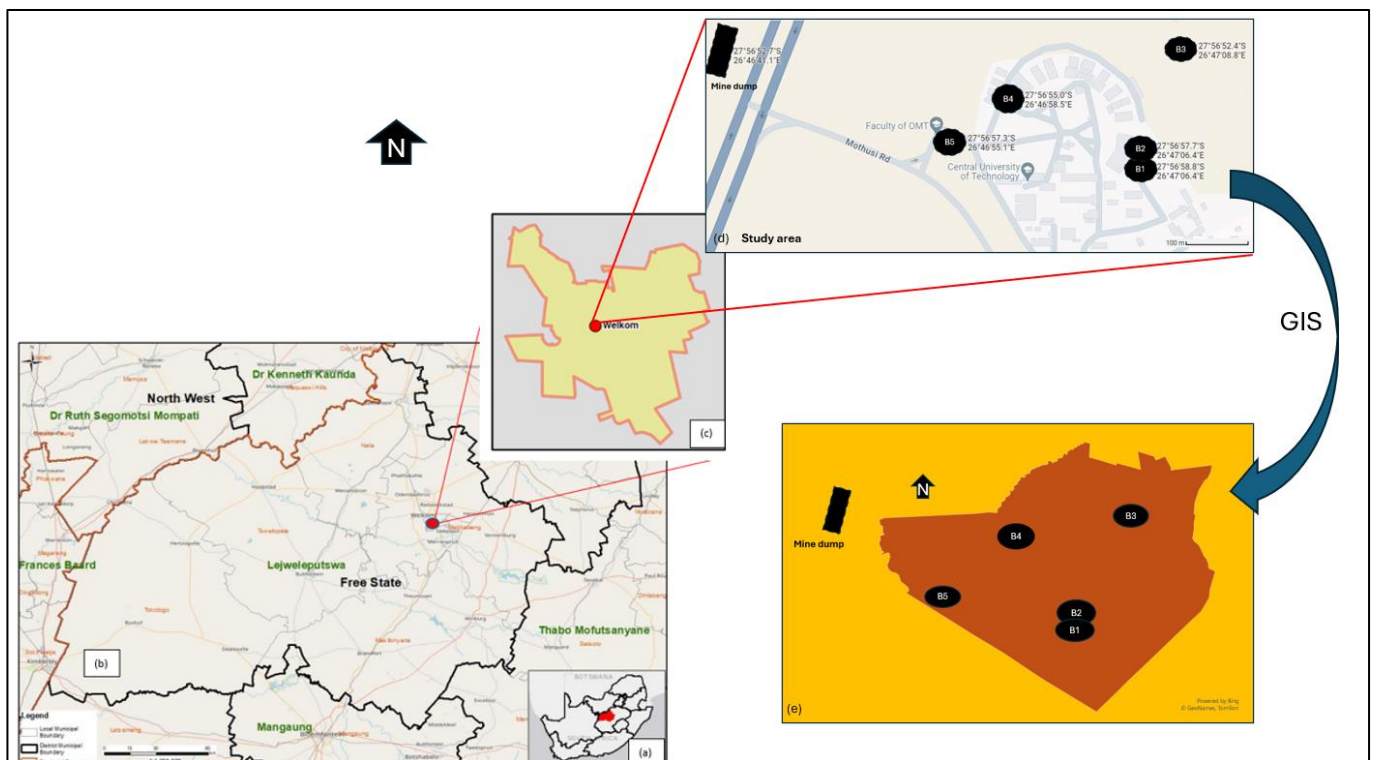


Figure 6.5 Study area map showing the inclusion of a GIS map in (e)

6.2 Assessing the groundwater flow direction

To determine the groundwater flow direction at the CUT Welkom campus, depth-to-groundwater data from five boreholes were mapped. The space between the boreholes was segmented to linearly interpolate the groundwater head values. Equipotential lines, representing equal groundwater levels, were plotted based on data from boreholes B2, B3, and B5, which had groundwater depths of 1.9 m, 2.7 m, and 2.3 m, respectively. The method for calculating the flow direction assumed the subsurface material had uniform hydraulic conductivity, allowing groundwater to flow perpendicular to the equipotential lines, from areas of higher to lower gradients (Cook, 2003).

6.2.1 Assessing the overall groundwater flow direction relative to the contaminant source

The groundwater flow direction between the boreholes as illustrated in Figure 4.1 was integrated with the groundwater flow direction chart developed in Chapter 5. An imaginary line perpendicular to the mine dump and borehole B5 was drawn to interpolate the direction of groundwater flow against the groundwater flow direction chart. Borehole B5 was used as it is the closest to the mine dump. Figure 6.7 shows the intricate details of the consolidated datasets on the map of the study area.



Figure 6.6 Groundwater flow direction relative to the groundwater flow chart and the mine dump

6.2.2 Classification and rating of the groundwater flow direction

Using the direction of the flow chart as indicated in Figure 6.2, the overall direction of groundwater flow was classified as positive. This is mainly because the direction of groundwater flow evaluated from boreholes B2, B3, and B5, was within the angle range of 0–90°, which lies within the first quadrant on the direction of the flow chart.

The positive rating of the groundwater flow direction across the five boreholes in the study site can thus be classified under the rating of 1 as shown in Table 6.2.

Table 6.2 Classification and rating of groundwater flow direction

Classification of groundwater flow direction	Positive flow (Direction of flow is away for the well) $0^\circ \leq \theta \leq 90^\circ$ ($270^\circ \leq \theta \leq -360^\circ$)	Undefined flow (Direction of flow is undefined) $\theta = x = 0$	Negative flow (Direction of flow is towards the well) ($90^\circ \leq \theta \leq 180^\circ$) ($180^\circ \leq \theta \leq 270^\circ$) $Y = 0$
Vulnerability class	Low vulnerability	Medium vulnerability	High vulnerability
Rating	1	3	5

According to the groundwater flow direction assessment conducted, the direction of groundwater flow is away from the boreholes and poses a low risk to the contamination of the groundwater at any of the five abstraction points. Figure 6.8 shows the map with the risk classifications of each borehole in relation to the groundwater flow direction from the mine dump.

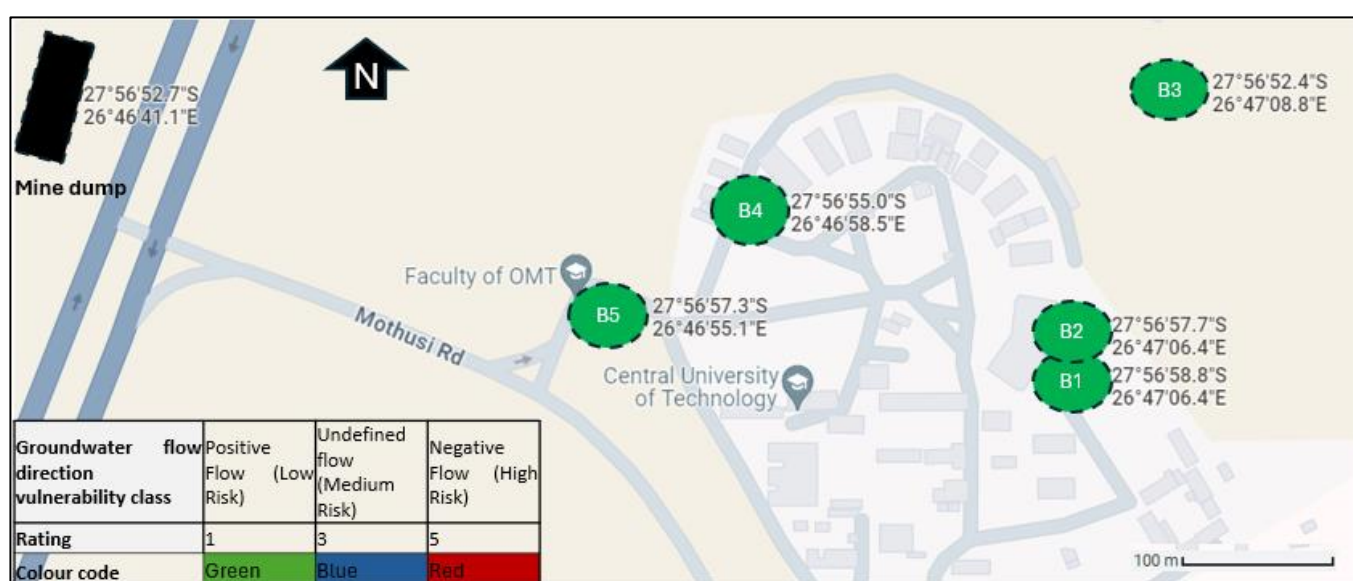


Figure 6.7 Impact of groundwater flow direction on contamination from mine dump

6.3 Water quality

The water quality parameter is critical to the DWAPH vulnerability assessment model as it gives an indication on the extent of groundwater contamination and hence vulnerability. The parameters to be tested included EC, pH, NO₃, SO₄, and Fe.

These five water quality parameters were selected as they are:

- very good and proven trace indicators of pollution in gold mine dumps (Kanda et al., 2021; Okereafor et al., 2019); and
- linked to chemical processes associated with gold mine dumps such as the degradation of pyrite into sulphuric acid and other associated compounds.

The WAI method was used for calculating the overall drinking water quality index of the five boreholes at the study site using the SANS 241 (2015) drinking water quality standards. Table 6.3 shows the average water quality parameter measurements across the five boreholes (B1–B5).

Table 6.3 Average water quality parameter values as measured from the five boreholes

Parameters	Limit Standard (SANS 241:2015)	Risk	B1	B2	B3	B4	B5	Mean (B1–B5)	Minimum	Maximum	Compliance %
EC	170 mS/m	Acute health	84.62	<14.7	202.31	62.75	138.89	122.14	<14.7	251.42	100
pH	5–9.7	Operational	7.04	7.55	6.80	7.64	6.81	7.17	6.81	7.55	100
NO ₃	11 mg/ℓ	Acute health	1.76	0.36	0.29	2.00	4.16	1.71	0.36	4.16	100
SO ₄	500 mg/ℓ	Acute health	134.85	1.11	146.51	99.40	89.36	94.25	1.11	375.46	100
Iron (Fe)	2 000 µg/ℓ	Chronic	154.1	132.5	141.21	133.4	137.81	139.8	132.5	154.1	100
	300 µg/ℓ	Aesthetic									

The overall water quality index was also calculated using the weighted arithmetic index method as per the SANS 241 (2015) drinking water quality standards. Table 6.4 shows the overall outcome of the drinking water quality index calculations.

Table 6.4 WQI calculations using the SANS 241 drinking water quality guidelines

Parameters	S _n	1/S _n	Sum (1/S _n)	K=1/(sum1/S _n)	Wi = K/S _n	Ideal value (V _o)	Mean conc Val (V _n)	V _n /S _n	V _n /S _n *100 = Q _n	W _n Q _n
pH	8.5	0.117647	5.21155615	0.191881268	0.022574267	7	7.17	0.11	11	0.248317
EC	1 000	0.001	5.21155615	0.191881268	0.000191881	0	2 051.42	2.05142	205.142	0.039363
NO ₃	11	0.090909	5.21155615	0.191881268	0.017443752	0	0.75	0.068182	6.818181818	0.118935
SO ₄	500	0.002	5.21155615	0.191881268	0.000383763	0	3075.46	6.15092	615.092	0.236049
Iron (Fe)	0.2	5	5.21155615	0.191881268	0.959406338	0	0.14	0.7	70	67.15844
		5.211556								67.80111

*S_n = Recommended standard value of the nth parameter

K = Proportionality constant

- V_n = Mean concentration value
 Q_n = Water quality rating
 W_n = Unit weight of each water quality parameter

An overall mean WQI of **67.8** was found.

The mean values for the individual boreholes B1–B5 are shown in Table 6.5.

Table 6.5 WQI values for boreholes B1–B5

Borehole sample	Water quality index	Water quality class
B1	74.5	Very poor
B2	64.5	Poor
B3	68.1	Poor
B4	65.3	Poor
B5	67.2	Poor
Mean(B1–B5)	67.8	Poor

The rating of the WQI was conducted using class parameters of the WAI method as shown in Table 6.6. Overall, the mean WQI was found to be in the poor rating and a rating of 3 was assigned.

Table 6.6 Water class parameters for the Weighted Arithmetic Index method

Water Quality Index	Water Quality Class	Rating
0–25	Excellent	1
26–50	Good	2
51–75	Poor	3
76–100	Very poor	4
Above 100	Not suitable for drinking purposes	5

The individual rating of the groundwater from each borehole was also rated using the classified water quality classes in Table 6.7. The water quality from boreholes B2–B5 was rated as poor, while the water quality from borehole B1 was rated as very poor.

Table 6.7 Rating of the water quality in individual boreholes

Borehole sample	Water Quality Index	Water quality class	Applied rating
B1	74.5	Very poor	4
B2	64.5	Poor	3
B3	68.1	Poor	3
B4	65.3	Poor	3

B5	67.2	Poor	3
Mean (B1–B5)	67.8	Poor	3

The poor rating of the drinking water meant that the water quality in the five boreholes was not of good drinking quality. Since the water quality is measured per point source, the applied ratings of each of the five boreholes were plotted on a map against the location of the mine dump. Figure 6.9 shows the water quality ratings and colour codes.

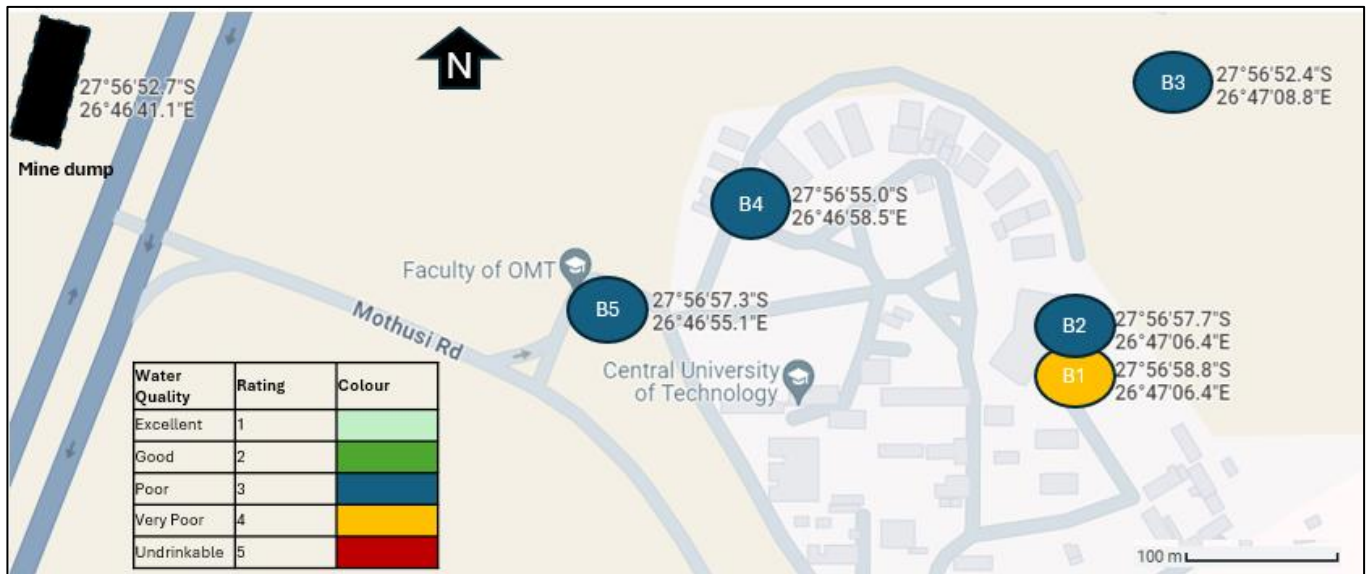


Figure 6.8 Water quality map with ratings and colour codes specific for each borehole

6.4 Aquifer rock type

The lithology of the Welkom study area was interpreted through the geological maps of South Africa. The aquifer media type found in the Welkom area is characterised by sandstone, shale, and limestone. Figure 6.10 shows the geological map of South Africa.

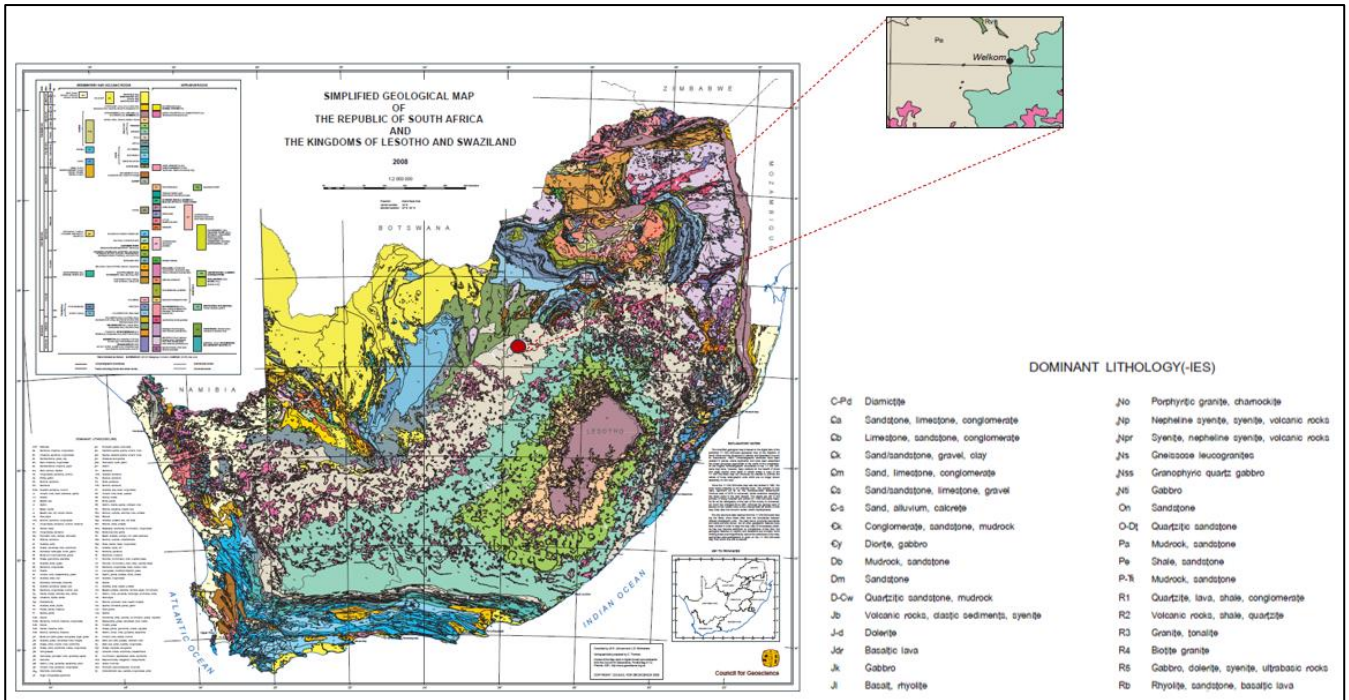


Figure 6.9 Geology map of South Africa (2008)

The aquifer rock type ratings were given a subjective range of ratings between 1 and 5 with the rating of sandstone, siltstone, and limestone being at 3 (Table 6.8). As the DWAPH method assumes that all five parameters have the same impact, no weight was assigned to the aquifer type rating.

Table 6.8 Adaptation of the aquifer rock types ratings

Rock type	Rating
Consolidated dense rock, shale, igneous and metamorphic rocks	1
Fractured igneous and metamorphic	2
Sandstone, siltstone, and limestone	3
Weathered basement	4
Unconsolidated sediment	5

In most instances, the aquifer media material is rated the same as the vadose material (Oke, 2020; Verlicchi and Grillini, 2020). The DWAPH model thus further assumes that the contaminant attenuation capacity of the aquifer depends not only on the aquifer media material but also on the amount and sorting of fine grain sediments within the vadose zone. The vadose zone and aquifer media structure are thus given the same magnitude of importance and can be used interchangeably.

The grain sizes of the rock type found within the Welkom area have larger grain sizes and porous openings with high permeability rates and low attenuation capacities. Figure 6.11 shows the aquifer rock type map of the study area which is dominated by sandstone, siltstone, and limestone.

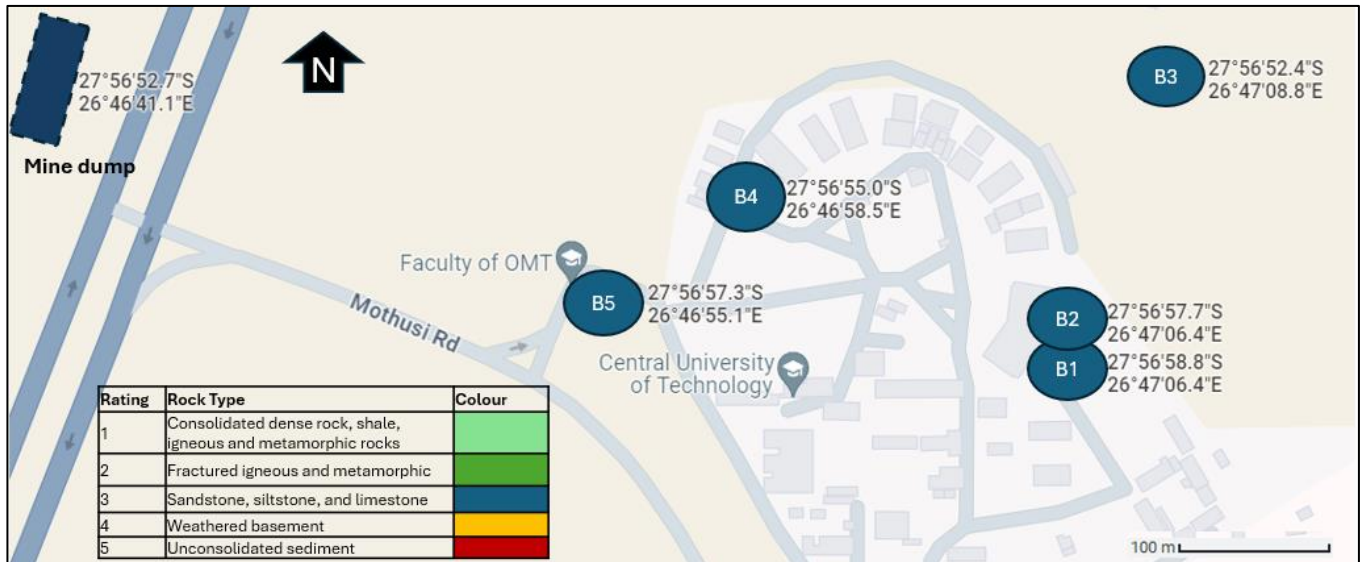


Figure 6.10 Aquifer rock type map underlying the boreholes and mine dumps in the study area

6.5 Precipitation

In the DWAPH model, precipitation represent the recharging force of the contaminants from the mine dump surface into the groundwater. The rainfall data of the Welkom area obtained from the SAWS between 1987 and 2023 were used for rating the rainfall parameter. The monthly rainfall data were averaged to find the annual rainfall data that were used for the DWAPH calculations. Table 6.9 shows the annual rainfall data between 1987 and 2023.

Table 6.9 Precipitation distribution between 1987 and 2023

LEGEND												
Total of the daily rainfall (in mm) by month												
*** indicates data is missing or is not yet available in the current month												
--- indicates that data is unavailable or was not requested												
= indicates that the average is unreliable due to missing daily values												
Monthly Daily Rain (mm) Data for station [0231279 9] Welkom												
Year	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC
1987	9	98.1	51.5	52.7	0	23	19.5	39.5	117.2	6.5	129.8	82
1988	27.5	484.5	212.3	176.1	0.5	19.6	3.5	0	80	73.5	75.5	58
1989	122.3	207.3	48.5	54.5	18	0	4	0	15.5	15	16.5	19
1990	16.5	87.5	106.2	63.8	0	18.5	0	20	0	0	9	65.5
1991	145	88.1	134.5	0	0	21	0	0	23	167.8	20.5	50.6
1992	0.3	12.5	26	10.5	0	0	1.1	30	0	58.5	109.5	0
1993	39	61.2	61.5	39	6	4.1	0	13.5	0	124.5	104.3	49.5
1994	166.2	224.6	59.7	6	0	0	5.5	0	0	0	30.6	15.2
1995	65.8	59.2	77.2	3	32.9	0	0	0	27.5	38.5	28.7	80.3
1996	99	133	67	38	0	1.6	36	12	7.5	12	137.3	127.5
1997	37.1	21.5	145.4	26	22	13.4	9	21.5	5	15	23	10
1998	109.1	124	124	22.1	0	0	6.5	6.5	18.9	46	75	50
1999	80.5	22	18.5	53.6	54.7	1.6	1.5	0	0	35	12	193
2000	103.8	82	116	52.8	17.5	8	0	0	62.9	21	93.5	85.5
2001	13.5	63.2	78.5	110	31	23	0.6	14.7	38.4	50.5	273.5	112.5
2002	60.5	22	20.3	25.5	0.5	8	0	96.3	12.5	8	5	82.7
2003	44	85.6	79.6	7	36	0	0	7	14	19	45.4	37.5
2004	32.6	78.2	164.5	120.5	0	0	0	16	22	5.5	7	31.5
2005	80.2	29	31	18	31.5	23	0	9	0	31.5	39	0
2006	263.1	130.5	128.8	76	31	0	0	129.2	11.5	56.7	94	28
2007	42	10.5	17.5	21.8	0	20	0	2	30	37.3	83	59.3
2008	105	76	79	1.7	52.5	35	0	0	0	6.5	164	48.1
2009	42.5	124.3	55	27	14.9	18	21.5	0	0	102.5	35.5	63
2010	119.3	115	103.5	39	8	9.9	0	0	0	0	76.2	108.9
2011	201	160	98	60	28.4	50.2	7	0	0	0	3.3	28
2012	29.5	9.5	39.2	17.5	0	7.5	12.6	0	1	25	18.3	17.5
2013	0	34.5	56.5	0	0	0	0	0	0	3.5	24	60.7
2014	52.2	61	33	16	0	0	0	4.8	0	15.4	14	
2015	0	0	0	0	0	0	0	0	0	0	0	0
2016		0.0=	0	51.5	30.5	9.5	13	0	0	0	0.1	13
2017	48	133.3	0	15	4.2	0	0	0	0	8.5	16.4	80
2018	70	44.5	51	39	0	0	0	7.5	12	137.3	127.5	7.5
2019	58	31	38	72	8	6	0	0	0	2	15	151
2020	43	11	78	45	3	1.6	36	12	0	9	23	35
2021	94	69	101	31	21	13.4	9	21.5	5	5.5	13	62
2022	8	48	37	24	20	0	6.5	6.5	18.9	13.9	135	8
2023	10	8	68	32	26	1.6	1.5	0	0	***	***	***

Source: SAWS (2023)

The average seasonal rainfall patterns for the summer and winter seasons were measured to assess the extent of rainfall received in the summer and winter seasons. The average rainfall received for the summer and winter seasons was 67.5 mm and 30.5 mm, respectively. Both rainfall patterns fell within the category rating of less than 200 mm.

Table 6.10 shows the precipitation rating scale of the DWAPH method. The average annual rainfall for the Welkom area was 38.1 mm and was assigned a rating of 1 as the rainfall is less than 200 mm.

Table 6.10 Precipitation rating scale

Rainfall (mm)	Rating
>500	5
400–500	4
300–400	3
200–300	2
<200	1

Figure 6.12 shows the precipitation map of the study area.

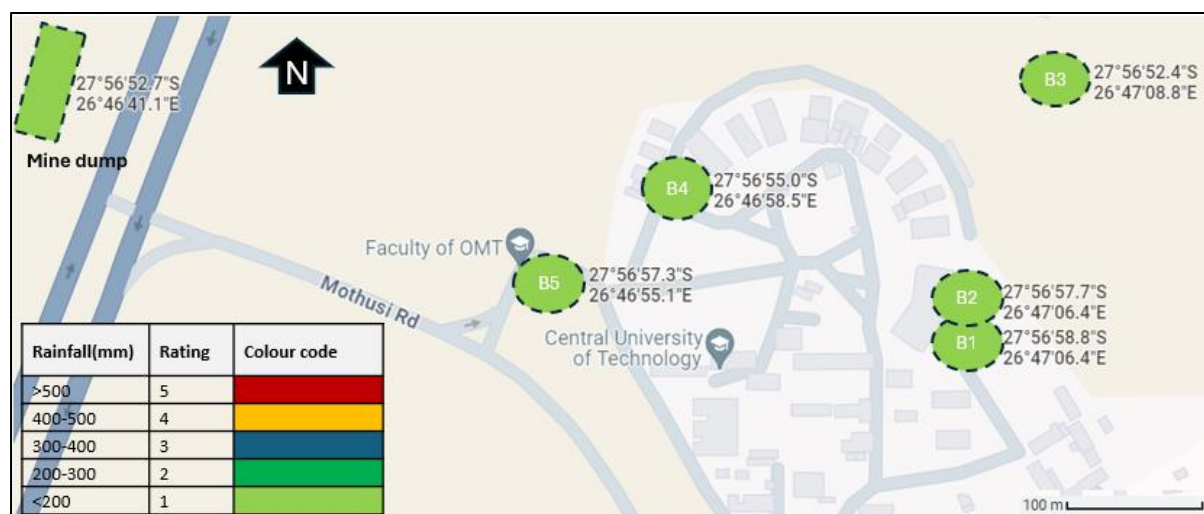


Figure 6.11 Precipitation/rainfall map of the study area aligned to specific point sources

6.6 Horizontal distance to contaminant source

The horizontal distance to contaminant source refers to the average distance between the contaminant source and the boreholes or target. In the context of this study, the mine dump was the contaminant source and the distance to contamination referred to the horizontal distance between the mine dump and the boreholes. The DWAPH method strongly relies on the effect of the horizontal distance on the contaminant.

The following steps were taken to determine the average horizontal distance between the mine dump and the boreholes:

Step 1: The distance between the mine dump and each of the five boreholes was measured using a Google Map scale. Figure 6.13 shows the measurements taken between the mine dump and each of the five boreholes.



Figure 6.12 Distance between the mine dump and the five boreholes located within the CUT Welkom campus

Step 2: The distances between the five boreholes and the mine dump were averaged to obtain a single value that could be used in the DWAPH methodology. Table 6.11 shows the average distances between the mine dump and the five boreholes.

Table 6.11 Average distances between mine dump and boreholes

Borehole	B1	B2	B3	B4	B5	Average
Distance from mine dump (m)	716.8	705.2	756.9	481.9	408.8	613.9

Step 3: The average distance to the contaminant source was 613.9 m and was assigned a rating of 5 as per the DWAPH distance ranges to contaminant ratings. The rating of 5 is the classification with the lowest distance to contaminant source and presents the highest distance vulnerability between the contaminant source and the borehole. Table 6.12 presents the distance to the contaminant table of the DWAPH method.

Table 6.12 Distance to contaminant source

Factor	Distance to contaminant source	Rate
Horizontal distance to contaminant (H)	0 – 1 000	5
	1 001 – 2 000	4
	2 001 – 3 000	3
	3 001 – 4 000	2
	≤4 001	1

The rating of each borehole relative to distances indicated in Table 6.12 is shown in Figure 6.14.

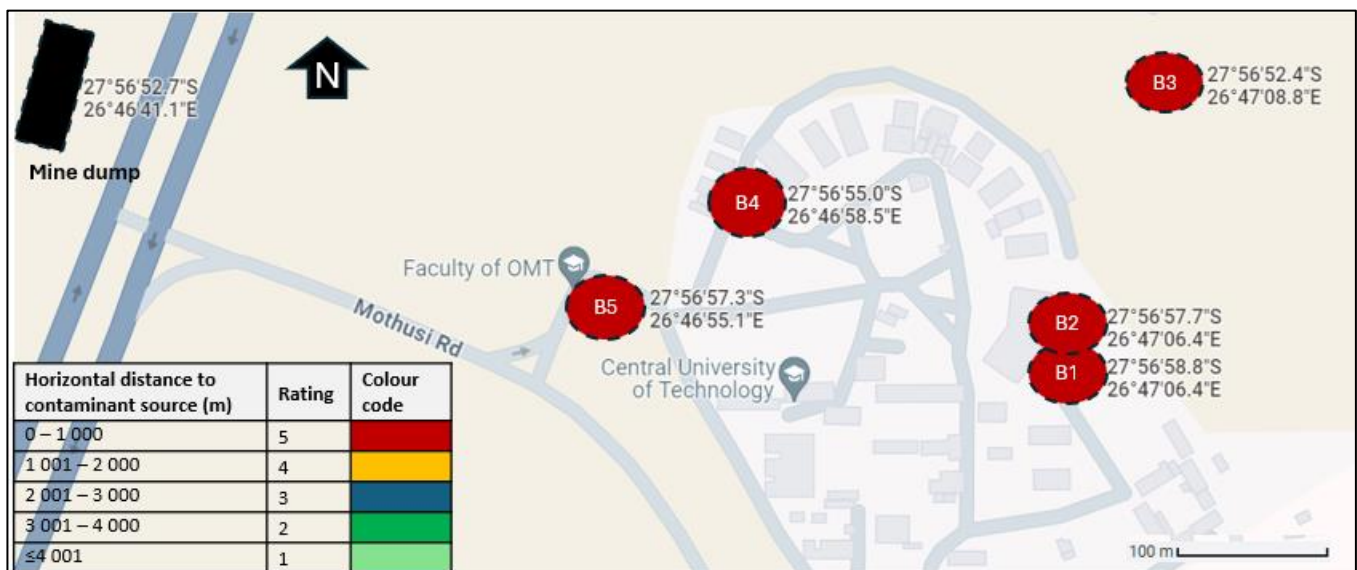


Figure 6.13 Specific horizontal distance to contaminant source

6.7 Calculation of the groundwater vulnerability assessment using the DWAPH method

All five parameters from the DWAPH vulnerability assessment method were assigned rating scales as shown in Table 6.13. The ratings to all parameters were objectively assigned using experimental and historical data. Because all the parameters were assumed to be contributing equally to groundwater vulnerability, none of the parameters were rated by weights.

The ratings highlighted in the darker shade of grey indicate the overall scores for each of the five parameters under the conditions of the study area.

Table 6.13 DWAPH model factors with the classification and ratings of the Welkom area

Number	Parameter	Class	Rate
1	Direction of groundwater flow (D)	Negative flow direction	1
		Undefined flow direction	3
		Positive flow direction	5
2	Water quality (W)	Excellent	1
		Good	2
		Poor	3
		Very poor	4
		Not suitable for drinking purposes	5
3	Aquifer type (A)	Consolidated dense rock, shale, igneous and metamorphic rocks	1
		Fractured igneous and metamorphic	2
		Sandstone, siltstone, and limestone	3
		Weathered basement	4
		Unconsolidated sediment	5
4	Precipitation (P)	>500	5
		400–500	4
		300–400	3
		200–300	2
		<200	1
5	Horizontal distance to contaminant (H)	0 – 1 000	5
		1 001 – 2 000	4
		2 001 – 3 000	3
		3 001 – 4 000	2
		≤4 001	1

Each of the parameter scales were assigned the following ratings as per the observed data:

$$\begin{aligned}
 \text{DWAPH}_{\text{index}} &= D_r + W_r + A_r + P_r + H_r \\
 &= 1 + 3 + 3 + 1 + 5 \\
 &= \mathbf{13}
 \end{aligned}$$

The DWAPH vulnerability index of 13 was classified under moderate vulnerability. Table 6.14 shows the vulnerability scale of the DWAPH method.

Table 6.14 Vulnerability scale for the DWAPH method

Vulnerability class	Low vulnerability	Moderate vulnerability	High vulnerability	Very high vulnerability
Range (x)	5–10	11–15	16–20	21–25

The outcomes from the overall DWAPH calculations were calculated and plotted on a vulnerability map of the study area. Figure 6.15 shows the overall groundwater vulnerability map for the study area.

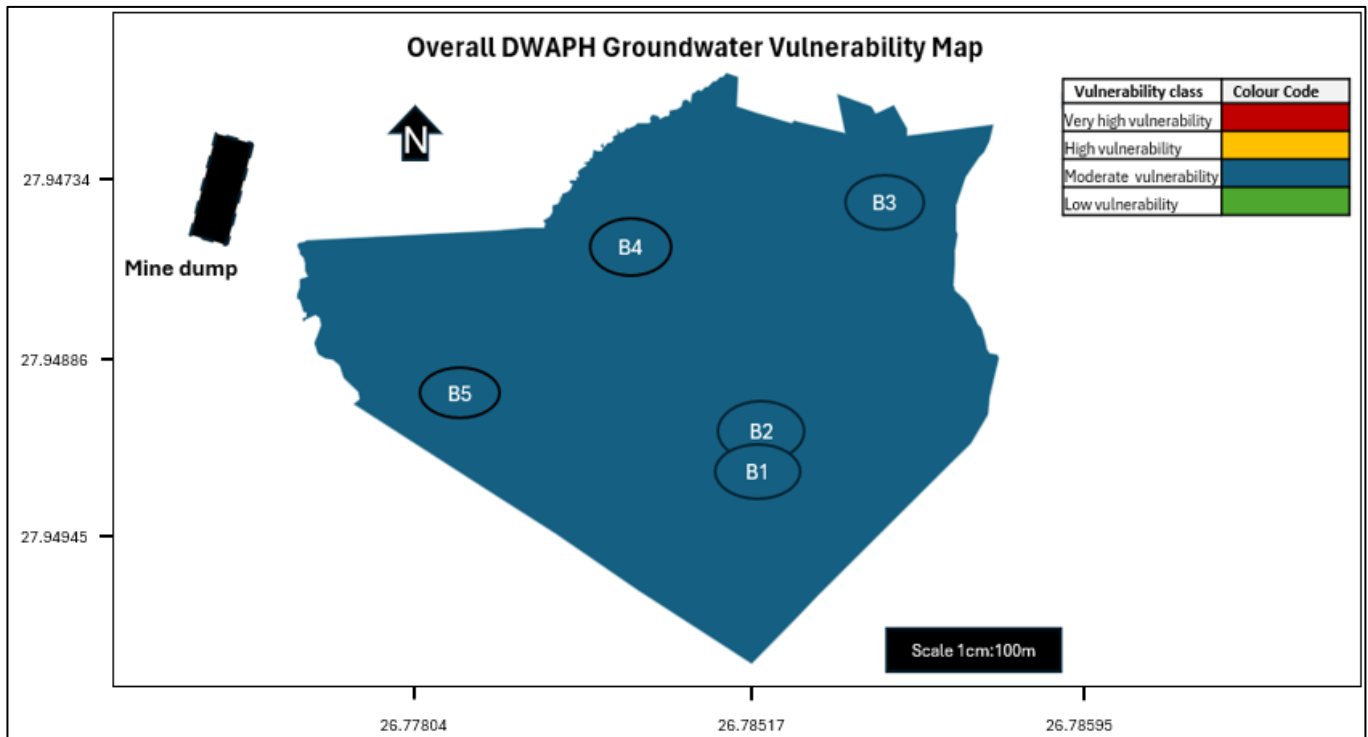


Figure 6.14 Overall DWAPH groundwater vulnerability map

6.8 Conclusion

The DWAPH groundwater vulnerability index for the study area was found to be 13, placing it in the moderate vulnerability category. This score is considered conservative, given that parameters such as water quality, distance to contaminant source, and precipitation, indicated a tendency towards higher vulnerability for the groundwater. The water quality index was estimated at 67.8, thereby falling in the "poor" category. The poor quality was largely due to elevated concentrations of sulphate (SO_4) and nitrate (NO_3^-), likely byproducts of the area's gold mining activities. The aquifer underlying the region is composed of sandstone, siltstone, and limestone, which are highly porous materials that significantly contribute to the increased vulnerability of groundwater to contamination. Additionally, the shorter distances between boreholes and the nearby mine dump were found to play a key role in elevating the groundwater's vulnerability. These factors collectively underscore the greater risk of contamination faced by the groundwater system in the Welkom area.

Chapter 7

Assessment of Groundwater Vulnerability in Welkom Using Other Conventional Groundwater Vulnerability Assessment Methods

7.1 Introduction

In Chapter 6, the groundwater vulnerability of the Welkom area was assessed by using the newly developed DWAPH vulnerability method. The outcome of the assessment indicated that the groundwater at the five boreholes assessed at the CUT Welkom campus area were moderately vulnerable to contamination from the mine dump that is located just under a kilometre from the boreholes. However, because the DWAPH vulnerability method was specifically developed and designed to assess the groundwater vulnerability of the Welkom and similar areas to mine dump contamination, there is a need to validate its reliability by drawing out comparisons with other vulnerability methods.

In this chapter, the vulnerability of the CUT Welkom campus study area to contamination from the mine dump was assessed by applying other conventional and established groundwater vulnerability assessment methods such as the GOD, AVI, and the RTt method. The selected vulnerability assessment methods were chosen because:

- their approach to assessing groundwater vulnerability differs from that of the DWAPH method and will present a balanced view on the performance of the DWAPH method;
- all three methods utilise few parameters in their calculations on groundwater vulnerability assessments; and
- each of the three methods are easy to compute and interpret, just like the DWAPH method.

Although the DWAPH method was developed for calculating specific vulnerability to contamination, it is expected that the GOD, AVI, and RTt methods may not produce very close outcomes as they were designed for assessing intrinsic vulnerability. The methods will, however, give an indication on the range of effectiveness of the DWAPH method.

7.2 Application of the GOD method

The GOD method is a convectional groundwater vulnerability assessment method that have been used to assess the groundwater vulnerability in coal and gold mining areas (Sukmawati Rukmana et al., 2020).

Originally, the GOD method used two basic parameters:

- The extent of accessibility of the aquifer saturated zone.
- The layers of contaminants lining saturated aquifers.

However, because of the lack of data, these parameters have been simplified to:

- hydraulic configuration of groundwater;
- overlay strata or vadose zone; and
- depth to groundwater

In this study, the groundwater vulnerability in the Welkom area was assessed using the following GOD equation:

$$\text{GOD index} = \text{Ca} \cdot \text{Cl} \cdot \text{Cd} \quad \text{Equation 7.1}$$

Where:

Ca = type of aquifer

Cl = type of lithology aquifer

Cd = depth of aquifer

The ratings and parameters are summarised in Table 7.1

Table 7.1 Parameters of the GOD method measured from the Welkom study area

Parameter	Classification using data at the Welkom study site	Rating	Comments
Type of aquifer	Unconfined (covered)	0.6	Literature review and dilution tests conducted indicated that the aquifer is fractured and unconfined, hence the rating is 0.6.
Type of lithology aquifer	Sandstone, siltstone, and limestone	0.7	Data from the geological map of South Africa indicates that the type of lithology aquifer at the Welkom study site is largely composed of sandstone, siltstone, and limestone, hence the rating of 0.7.

Depth to groundwater	The average depth to groundwater value for the five boreholes = 2.32 m Hence, 2.32 m ≤ 5 m	0.9	The average of the measured depth to groundwater values of the five boreholes across the summer and winter seasons was 2.32 m, which fits into the depths to groundwater values of ≤ 5 m.
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Equation 7.1 shows the overall GOD vulnerability score:

$$\text{GOD index (Aquifer pollution vulnerability)} = 0.6 \times 0.7 \times 0.9 = \mathbf{0.38} \quad \text{Equation 7.2}$$

Thus, the aquifer pollution vulnerability is in the range $0.3 \leq x \leq 0.4$ and classified as moderate.

7.2.1 Comparison of the DWAPH and GOD vulnerability scores

The moderate vulnerability score identified through the GOD method was equally comparable to the moderate vulnerability score obtained through the DWAPH method. Because the DWAPH method utilises four scale classes ranging from very low to very high vulnerability, and the GOD scale utilises five scales ranging from negligible to extreme vulnerability, the range for the DWAPH moderate class was slightly wider than that of the GOD method. However, both methods were able to arrive at the same conclusion on their assessment of the Welkom groundwater vulnerability. Figure 7.1 shows a comparison of the DWAPH and GOD vulnerability assessment scales and Figure 7.2 shows the comparison of the DWAPH and GOD groundwater vulnerability assessment maps respectively.

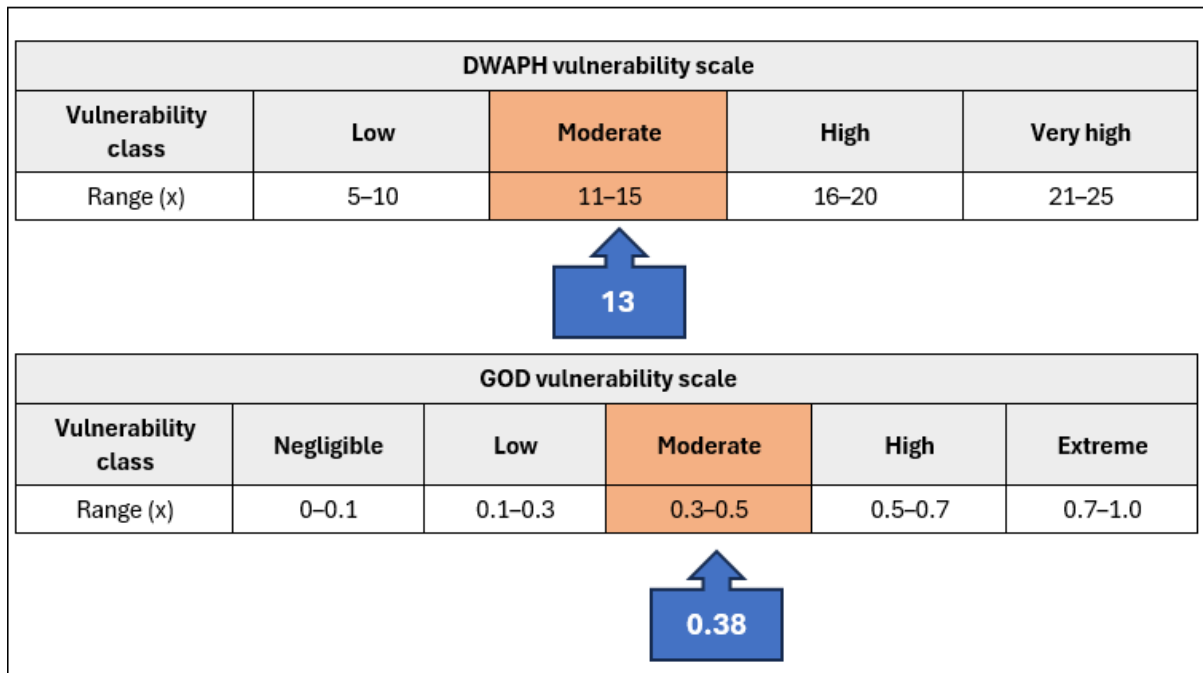


Figure 7.1 Comparison of the DWAPH and GOD vulnerability scales on Welkom groundwater vulnerability

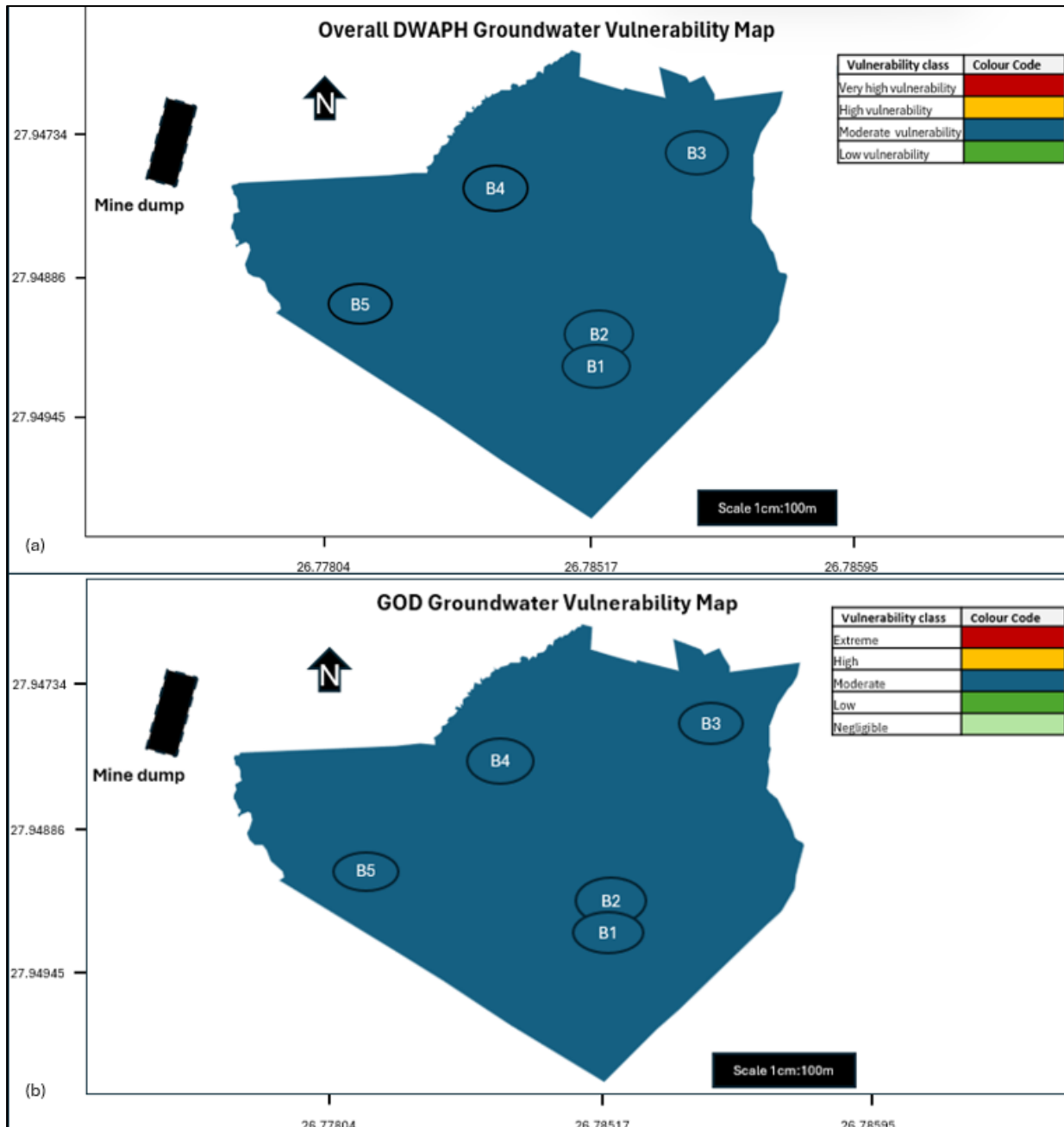


Figure 7.2 Comparison of the (a) DWAPH and (b) GOD groundwater vulnerability assessment methods

7.3 Application of the AVI method

The AVI method applies two parameters: the rock thickness above the aquifer (d), and the hydraulic conductivity from the rock. The hydraulic resistance (c) can thus be calculated as follows:

$$c = \sum_{i=1}^n \frac{d_i}{K_i} \quad \text{Equation 7.3}$$

Where:

- c = Hydraulic resistance
- i = Value of the n^{th} parameter
- n = Number of sedimentary units above the aquifer
- d_i = Thickness of the vadose zone
- K_i = Hydraulic conductivity of each protective layer (cm/s)

Table 7.2 shows the calculation of hydraulic resistance for the AVI method.

Table 7.2 Summary of the calculations and outcomes on hydraulic resistance

Layer	Thickness of vadose zone (d)	Hydraulic conductivity (K)	d/K
1	10	604	0.016556
2	20	1 650	0.012121
3	25	1 776	0.014077
4	35	1 811	0.019326
5	40	1 850	0.021622
$c = \sum(d/K)$			0.083702
Log @			-1.08

The thickness and lithology of the aquifer were determined using data from electrical sounding studies in existing literature. Hydraulic conductivity was assessed through on-site dilution tests. Hydraulic resistance values reflect the rock's capacity to transmit groundwater within a restricted volume, also indicating the estimated time for contaminants to traverse from the overlaying lithology to the groundwater. These resistance values serve as indicators of groundwater vulnerability. Table 7.3 illustrates the correlations and vulnerability classifications within the AVI method.

Table 7.3 AVI scale relationship between hydraulic resistance value and groundwater vulnerability

Hydraulic resistance	Log (Hydraulic resistance)	Vulnerability level
0–10	<1	Very high
10–100	1–2	High
100–1 000	2–3	Moderate
1 000–10 000	3–4	Low
>10 000	>4	Very low

Since the hydraulic resistance was 0.084 and its logarithmic equivalent was –1.08, the extent of groundwater vulnerability is classified as very high.

7.3.1 Comparison of the DWAPH and AVI vulnerability scores

While the DWAPH vulnerability method classified the vulnerability of the Welkom area to contamination from the mine dump as moderate, the AVI classified the vulnerability of the Welkom area to contamination as very high. This can be attributed to the high hydraulic conductivity values identified across the layers of sediment within the aquifers. Moreover, unlike the DWAPH method which considered at least five parameters, the AVI method utilised various assumptions:

- The contaminant is always released on land.
- Contaminants behave in the same way as water, thereby minimising the role of diffusion, advection, and other solute transport mechanisms.
- Land use and seasonal factors are not considered (Ducci and Sellerino, 2022; Oke et al., 2016).

Despite these assumptions and limitations, the outcomes from the AVI method cannot be completely disregarded as they assist in providing the structural implications of aquifers to groundwater vulnerability, which many other methods, including DWAPH, do not adequately cover for. Figure 7.4 shows a comparison between the DWAPH and AVI vulnerability methods.

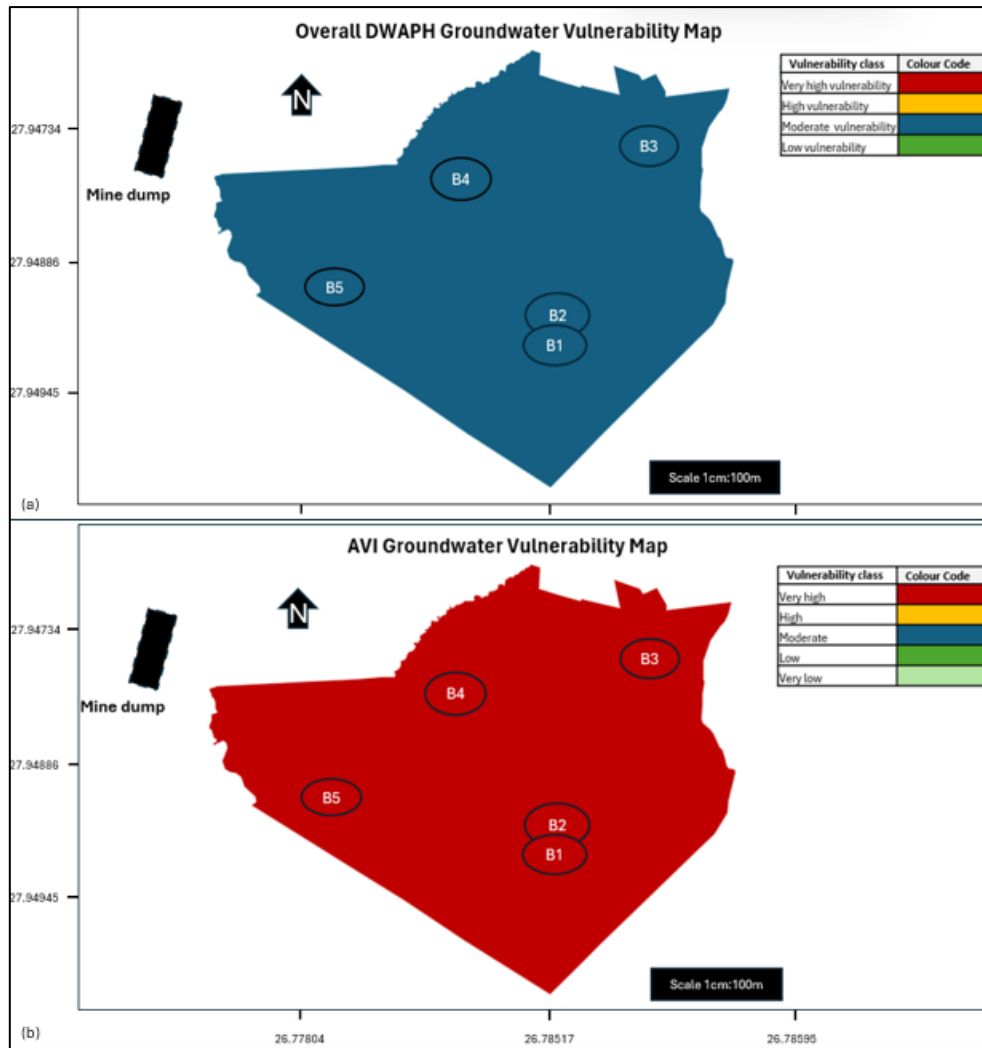


Figure 7.3 Comparison of the (a) DWAPH and (b) AVI groundwater vulnerability assessment methods

7.4 Application of the rainfall travel time method

To further validate the impact of the DWAPH method in the Welkom area, the rainfall travel time (RTt) method was also applied to context. The RTt method was developed by Oke (2015) to evaluate the vulnerability of shallow groundwater aquifers in the Dahomey basin in western Nigeria. The concept of the RTt method is mainly based on the hazard–pathway–receptor model. The RTt is mainly a subjective vulnerability method that considers rainfall, slope, porosity, hydraulic conductivity, and depth to water table as the main parameters affecting groundwater vulnerability. However, due to the limitations in the parameters and data used in the RTt method, it can only be applied to regions with very limited data availability. Oke (2015) recommended that the RTt method should always be used in conjunction with other methods to compensate for its weaknesses.

7.4.1 Rainfall

The RTt method considers rainfall as the main driving force for the transporting of contaminants into the groundwater through infiltration. The average annual rainfall received in the Welkom area is 38.1 mm. This rainfall amount is very low and was assigned a weight of 1, according to the RTt classification. Table 7.4 shows the rainfall classifications as per the RTt method.

Table 7.4 Rainfall classification as per the RTt method

Rainfall (mm)	Weight
>1 600	5
1 200–1 600	4
800–1 200	3
400–800	2
<400	1

7.4.2 Depth to water

The depth to groundwater parameter is important as it determines the vertical distance in which contaminants need to travel before reaching the groundwater. Just as in the horizontal distance, the longer the depth to groundwater distance, the less chances of contaminants being transported to the groundwater. The average depth to groundwater value for the five boreholes in the CUT Welkom campus was 2.32 m. Table 7.5 shows the classification of the depth to groundwater as per the RTt method.

Table 7.5 Classification of depth to water as per the RTt method

Range (m)	Weight
0–5	5
5–10	4
10–25	3
25–50	2
>50	1

The depth to groundwater was classified as very shallow within a range of 0–5 m and hence given a weight of 5. The shallowness of the groundwater poses a great risk of contamination as per the RTt scale.

7.4.3 Slope

The slope of an area is usually defined as the difference between the highest and lowest topographic points of a study area. The slope is an important aspect of groundwater vulnerability as it has effects on runoff, infiltration, and recharge of an area. In the context of this study, the slope between the mine dump and the location of the boreholes has a great effect on how contaminants migrate from the mine dump to the underlying aquifer and, hence, the boreholes.

An assessment of the slope in the study area measured against the highest peak on the mine dump against the borehole location indicated a slope of 1.02 m. According to the weights assigned by Oke (2015), the slope in the range of 0–2 m has a weight of 2.0. This means this area encourages a very low amount of run-off and, hence, higher infiltration, a condition which is well suited for aiding contamination of the groundwater and, hence, supporting greater vulnerability. Figure 7.4 shows the elevation of the mine dump slope relative to the CUT campus.



Figure 7.4 Elevation of the mine dump slope relative to the CUT Welkom campus boreholes

Table 7.6 shows the range of slope data together with the assigned weights as per the RTt method.

Table 7.6 Range of slope and assigned weight

Slope (m)	Weight
0–2	2.0
2–5	1.6
5–25	1.2
25–50	0.8
>50	0.4

7.4.4 Porosity

According to Oke (2015), the porosity of a geological material is largely affected by the size of particles within an area. In the RTt rating and weighting system, the porosity of soil particles ranged between 0 and 1, with soil ratings closer to 0 having very low porosity and soil particles close to 1 having higher porosity and, hence, faster infiltration rates. Table 7.7 shows the classification of the various levels of porosity and soil type as suggested by Oke (2015).

Table 7.7 Classification of porosity as per the RTt method

Porosity	Soil type	Weight
0.40–0.70	Clay	5
0.35–0.50	Silt	3
0.25–0.50	Sand	1
0.25–0.40	Gravel	1

Source: Freeze and Cherry 1979, cited in Oke (2015)

The porosity of the soil at the Welkom study site was classified under silt and given the weight of 3.

7.4.5 Hydraulic conductivity

The movement of groundwater controls the movement of contaminants. A strong relationship exists between the hydraulic conductivity of soil and the soil textural properties. While soil texture affects the movements of groundwater, the grainsize of soil particles also plays a major role in the movement and transport of water (Daly et al., 2002). The smaller the soil grains are, the lower hydraulic conductivity they have, the

lower the amount of water flow is allowed. Table 7.8 shows the soil textural classifications and hydraulic conductivities for classifying the RTt method.

Table 7.8 Soil textural classifications and hydraulic conductivity

Hydraulic conductivity	Soil texture	Percentage of clay	Weight
$10^{-6} - 10^{-14}$	Clay	60%	5
$10^{-2} - 10^{-11}$	Sandy clay and silty clay	40–50%	4
$10^{-3} - 10^{-8}$	Silty clay loam, clay loam, sandy clay loam	30–40%	3
$10^{-1} - 10^{-5}$	Silty loam, sandy loam, and loamy sand	10–20%	2
$100 - 10^{-4}$	Sand and silt	10%	1

The soil textural classification and hydraulic conductivity measurements conducted at the Welkom study area had properties of silty loam and silty sand soil and was given a weight of 2.

7.4.6 Rainfall travel time vulnerability class

The classification of the RTt method is based on a vulnerability index with a scale ranging from a minimum value of 12 to a maximum value of 100, with five vulnerability classes. Table 7.9 shows the rating of the RTt vulnerability classes together with corresponding RTt indices.

Table 7.9 Classification of the rainfall travel time vulnerability class and index

RTt index	Vulnerability class
12–29	Very low
29–47	Low
47–65	Moderate
65–83	High
83–100	Very high

The RTt value is calculated by substituting the weighting of the parameter values into the equation:

$$RTt = R \times 10 + (D \times S) / (K_{sat}/\theta) \times 10 \quad \text{Equation 7.4}$$

Where:

- R = Rainfall rating
- D = Depth to groundwater
- S = Slope

K_{sat} = Hydraulic conductivity

θ = Porosity

$$RTt = 1 \times 10 + (5 \times 2) / (3) \times 10 = \mathbf{43.33}$$

Equation 7.5

The substitution of the weights into the RTt equation yielded a value of 43.33. According to the RTt classifications, the range of 29–47 falls in the low vulnerability class. The low vulnerability index is an indication that factors such as precipitation, geology, topography, and hydraulic activity of the Welkom area do not overly expose the groundwater to contamination within the study area. It should, however, be noted that the groundwater vulnerability assessed through the RTt method is an intrinsic type of vulnerability and may lack precision in directly addressing specific vulnerability in relation to the mine dump.

7.4.7 Comparison of the DWAPH and rainfall travel time scores

While the RTt method utilised a five-point scale ranging from a very low vulnerability class (12–29) to a very high vulnerability class (83–100), the DWAPH method utilised a four-point scale ranging from low vulnerability (5–10) to very high vulnerability (21–25) classes. This means that the characteristics identified in the low vulnerability classes of the DWAPH method are encompassed with the *very low* and *low* scales of the DWAPH vulnerability index classes.

The moderate vulnerability scores identified by the DWAPH method is a representation of specific vulnerability as it considered groundwater direction, distance, and water quality from the contaminant source, whereas the DWAPH method focused on specific vulnerabilities and therefore, the very low vulnerability score is a representation of intrinsic vulnerability.

Figure 7.5 shows a comparison of the DWAPH and RTt method vulnerability maps.

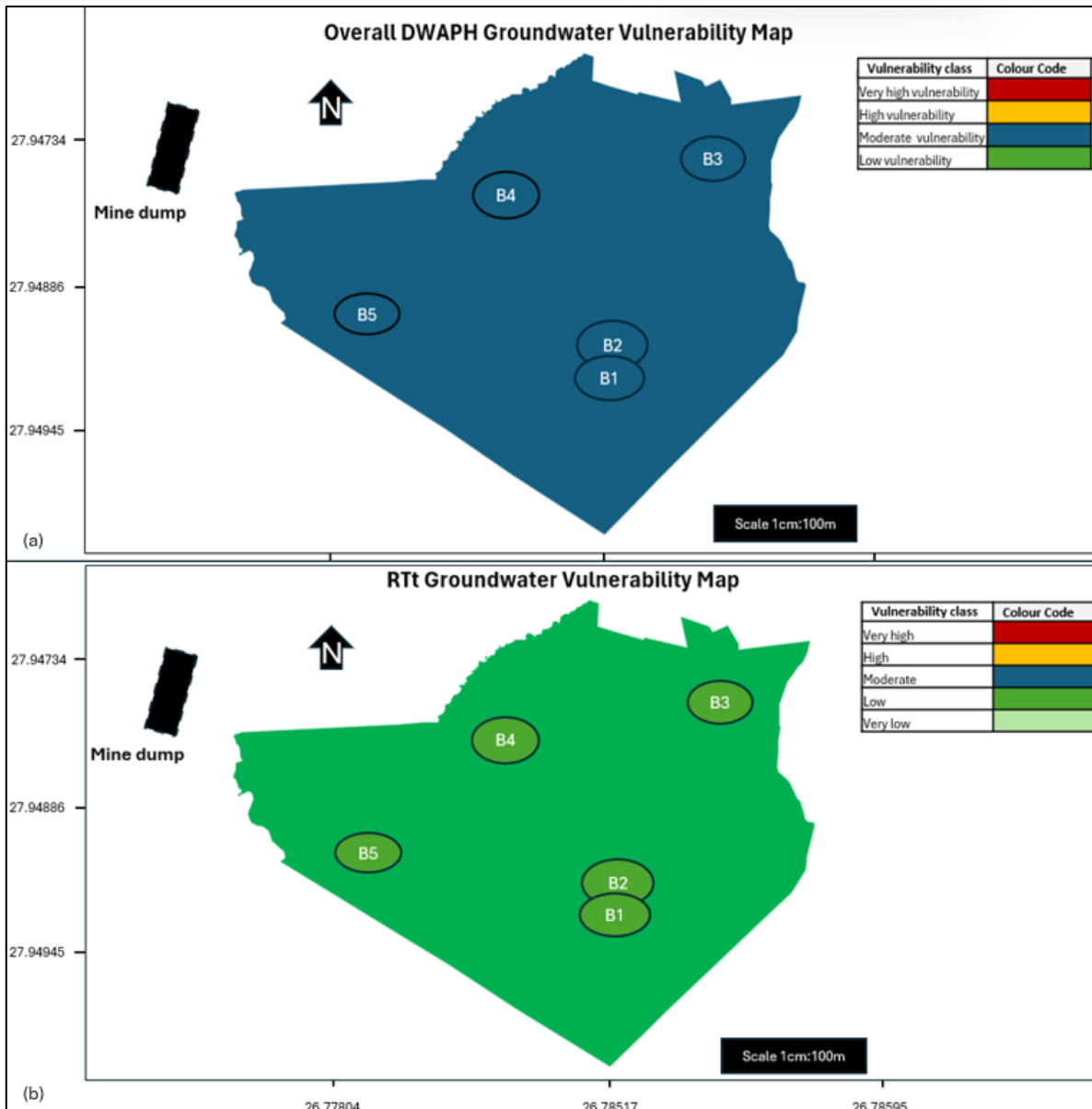


Figure 7.5 Comparison of the (a) DWAPH and (b) RTt groundwater vulnerability assessment methods

7.5 Overall comparison of the DWAPH and common existing methods

The DWAPH vulnerability assessment method is a subjective method designed to assess the impact of mine dumps on groundwater vulnerability. The groundwater vulnerability scale of the DWAPH method is composed of a five-point scale that moves from low vulnerability to very high vulnerability. To assess its reliability, the parameters and scales from the DWAPH method were compared to other existing groundwater vulnerability assessment methods. The selected methods included the GOD, AVI, and RTt methods. Table 7.10 shows the criteria and parameters for each of the methods chosen.

Table 7.10 Parameters considered under the DWAPH, GOD, AVI, and RTt methods

Factor/Parameter	DWAPH	GOD	AVI	RTt
Top soil thickness	-	-	-	+
Topsoil texture	-	+	-	+
Top soil structure	-	+	-	-
Rock/Subsoil permeability	+	+	+	+
Rock/Subsoil thickness	+	+	+	+
Depth of the unsaturated zone	+	+	+	+
Distance to possible contaminant source	+	-	-	-
Fracturing	+	-	-	+
Geomorphological features	+	+	+	-
Travel time estimation	-	-	-	+
Confined condition	+	-	-	-
Direction of groundwater flow	+	-	-	-
Slope gradient	+	-	-	+
Land use	+	-	-	-
Recharge	+	-	-	+
Hydraulic characteristics of aquifer	+	+	-	-
Water quality	+	-	-	-
Resources vulnerability	+	+	+	+
Source vulnerability	-	-	-	-

Compared to the other three methods, only the DWAPH method considered the most parameters, followed by the RTt, GOD, and AVI methods, respectively. Because of its specificity, only the DWAPH method considered the use of the land use, ground flow direction, and water quality parameters among all four methods. All four vulnerability assessment methods considered resource vulnerability and were not designed to be applied on source vulnerability assessments.

The range of each of the four vulnerability classes were compared against a five-scale blanket rating, ranging from very low vulnerability to very high vulnerability. Table 7.10 shows how each of the vulnerability scales were distributed for each of the four methods. Compared to the other three methods, only the DWAPH method had a four-scale distribution, ranging from low vulnerability to very high vulnerability. Moreover, the DWAPH method is the only scale that deals with specific vulnerability assessments as compared to the other three scales.

Table 7.11 compares the vulnerability classes and ranges of the four methods.

Table 7.11 Comparisons of vulnerability classes and ranges

Method \ Class	Very low vulnerability	Low vulnerability	Moderate vulnerability	High vulnerability	Very high vulnerability
	Green (0–20%)	Yellow (21–40%)	Blue (41–60%)	Purple (61–80%)	Red (81–100%)
DWAPH	–	5–10	11–15	16–20	21–25
AVI	$\text{Log } c > 4$	$3 \leq \text{Log } c \leq 4$	$2 \leq \text{Log } c \leq 3$	$1 \leq \text{Log } c \leq 2$	$\text{Log } c < 1$
GOD	0–0.1	0.1–0.3	0.3–0.5	0.5–0.7	0.7–1.0
RTt	12–29	29–47	47–65	65–83	83–100

7.6 Conclusion

In this chapter, the three groundwater vulnerability assessment methods, the GOD, AVI, and RTt, were applied to evaluate the extent of groundwater vulnerability in the Welkom area. The objective was to compare the outcomes of their assessments with the results from the DWAPH method. Despite all three methods utilising the same datasets, their outcomes were not the same due to a variety of factors that were mainly based on their design. The GOD method identified the extent of vulnerability of the Welkom area as moderate, while the AVI and the RTt methods identified the extent of groundwater vulnerability in the Welkom area as very high, and low respectively. This can be attributed to the difference in approaches as all three methods focused on different sets of parameters. Only the GOD method had a direct correlation to the DWAPH method. Aside from differences in water quality, both the GOD and DWAPH methods provide straightforward approaches by assessing a limited number of parameters, three in the case of GOD and five for DWAPH. The similarity in the results from these methods can be attributed to the fact that both rely on parameters that are well-suited to the specific environmental conditions under study. By focusing on key indicators relevant to the local context, both methods ensure that their assessments are not only comprehensive but also tailored to the unique characteristics of the environment, leading to comparable outcomes. It should however be noted that all three methods, compared, were designed for assessing intrinsic and not specific vulnerability. Further validation of these groundwater vulnerability assessment techniques together with the DWAPH method will be conducted in the next chapter.

Chapter 8

Validation and Significance of the DWAPH Method

8.1 Introduction

The validation of groundwater vulnerability is a process whereby the groundwater vulnerability indices are verified or compared with the measured, biological, chemical, or physical variables of a study. Just as the DWAPH vulnerability method was compared to other methods, it is also critical for the DWAPH vulnerability method to be compared against independent variables to ensure its validity. It is thus important to determine the ability of the DWAPH method to find values that give a close reflection of the physical environment.

For this study, it is important that a comparison between the projected groundwater vulnerability index value be compared to some of the variables and indicators for pollution. Several methods can be applied for the validation of vulnerability assessments. The most common validation methods include the use of hydrographs, chemographs, bacteriology, analogy studies, and artificial and natural tracers (Abdullah, 2020; Paun et al., 2016). The validation of the DWAPH method in this study was conducted using the NO_3^- and bacteriological counts methods. This is because NO_3^- concentration can be a useful pollution indicator to recognise the changes and evolution of water quality. Moreover, the availability of data for NO_3^- and bacterial concentrations makes it easy to apply.

8.2 Validation against nitrate

The validation of the DWAPH vulnerability method was developed by comparing NO_3^- concentration in groundwater to the groundwater vulnerability index. The NO_3^- concentration was used as the proxy for contaminant measure because of its naturally low presence in groundwater as well as its ability to move at great depths (Maqsoom et al., 2020). Furthermore, the use of NO_3^- has been applied for validating results from many groundwater vulnerability studies due to its high solubility. Some of the studies that utilised NO_3^- level for validation of vulnerability models include Li et al. (2021), Maqsoom et al. (2020) and Moges and Dinka (2021).

The validation of NO₃⁻ was conducted by assessing the concentration of NO₃⁻ across the five boreholes at the study site against their vulnerability indices. The NO₃⁻ values were obtained from the average of the summer and autumn seasons values between 2022 and 2023. Low NO₃⁻ values with concentrations ranging between zero to just over 1 mg/l were observed for the dry season while for the wet season, the concentration significantly rose to just over 4 mg/l.

For validation purposes, the DWAPH method was modified to estimate the specific groundwater vulnerability due to the mine dump. During the modification, the parameter system of the DWAPH method was optimised by adjusting the original parameter of horizontal distance to groundwater. The DWAPH method was applied to the study area and the groundwater vulnerability index was established. Table 8.1 shows the optimised DWAPH parameter against the original DWAPH parameter.

Table 8.1 Adjusted DWAPH parameter on horizontal distance to contaminant

Horizontal distance to contaminant (m)		Rating
Original DWAPH parameter	Optimised DWAPH parameter	
0–1 000	0–200	5
1 001–2 000	201–400	4
2 001– 3 000	401–600	3
3 001–4 000	601–800	2
≤4 001	≤ 801	1

The optimised DWAPH index outcome was plotted against the average NO₃⁻ concentration to establish the extent of the relationship between the two variables. Figure 8.1 shows the plot of the DWAPH vulnerability indices against NO₃⁻ concentrations for each of the five boreholes.

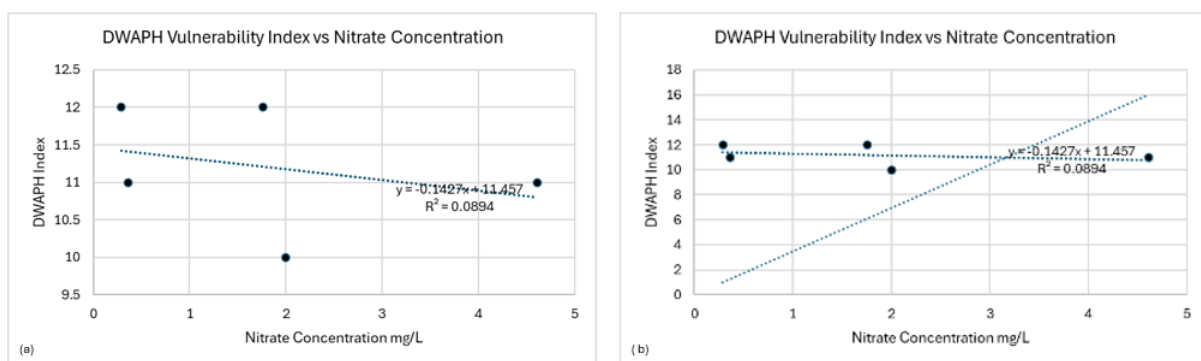


Figure 8.1 Regression plots of the DWAPH index against nitrate concentrations (a) original and (b) with intercept

The correlation of NO_3^- and the DWAPH model showed that $R^2 = 0.0894$. Although this correlation was not very strong, the validation results indicate that with adequate data, the DWAPH model can be used for estimating groundwater vulnerability with modest outcomes. In most cases, the adjustments of the parameters based on the context helps in optimising the performance of the DWAPH model.

8.3 Validation with bacteriology

Further validation of the DWAPH vulnerability assessment method was conducted using bacteriological data. It is difficult to test the water quality for all pathogens related to waterborne diseases because of the complexity of the testing and the time and cost related to it (Ashbolt et al., 2001). Total coliforms and *E. coli* were therefore used as indicator organisms for possible groundwater contamination in this validation study. This is because it is relatively easy and inexpensive to test for coliform bacteria compared to other methods such as borehole pumping tests. Figure 8.2 shows the different classes of coliform bacteria applied in this study.

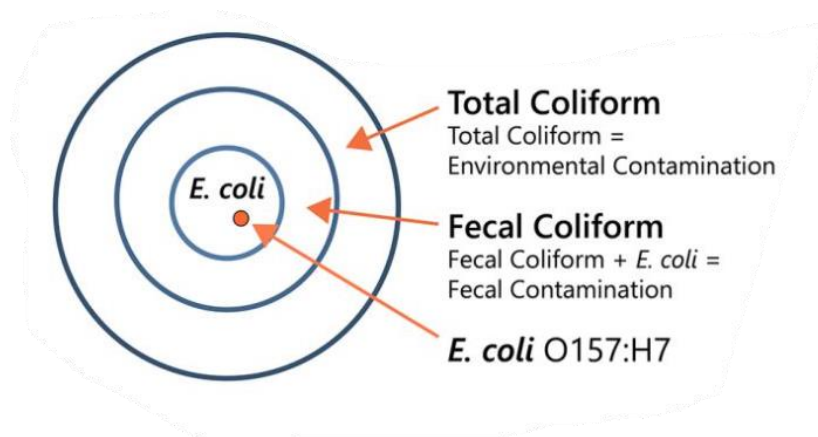


Figure 8.2 Classes of coliform bacteria used as indicator organisms (Water Research Commission, 1995)

Coliform bacteria concentrations in groundwater is affected by interactions between biological, chemical, and physical parameters (Dzwaairo et al., 2006). Seo et al. (2019) found relationships between coliform bacteria and water quality factors such as pH, EC, and biological oxygen demand at weir stations in the Nakdong River in South Korea.

The WHO (2022) stipulates that an ideal indicator organism should fulfil several criteria, such as:

- It ought to be detectable when the pathogen is present and undetectable in unpolluted water.
- It's concentration should surpass that of the pathogens it signifies.
- It's endurance in the environment and resilience to treatment procedures should mirror that of pathogens.
- It must not pose any risk to human health.
- It should be readily identifiable and isolatable.

However, at present, there is no absolute indicator that complies with all the above criteria. Table 8.2 shows some of the coliform indicators used to track drinking water quality in water treatment plants and their possible reasons for detection.

Table 8.2 Coliform indicators for drinking water quality

Microbial indicator	Possible reasons for detection of indicators	Most acceptable concentration or baseline level
<i>E. coli</i>	Recent faecal contamination, possible presence of pathogenic organisms	Non-detectable per 100 mℓ
Faecal coliforms	Inadequate treatment and disinfection, bacterial regrowth, or infiltration in the distribution system.	Non-detectable per 100 mℓ in treated water.
Total coliforms	Indicates that the water may be vulnerable to contamination and experiences bacterial growth	Non-detectable for 100 mℓ in treated water
Heterotrophic plate count	Indicates poor water quality that experiences bacterial growth	Depends on the system and site-specific conditions

Source: Verhille (2013)

8.3.1 Comparison of risk profiles for total coliforms and *E. coli* against the DWAPH indices

The vulnerability index for each borehole was calculated using the DWAPH vulnerability assessment method with adjustments of the scale on the horizontal distance parameter to optimise the outcome as shown in Table 8.1. The vulnerability of each of the five boreholes were then calculated and compared to the concentrations of total coliforms and *E. coli* collected in the summer season between 2022 and 2023. The total coliforms and *E. coli* were classified under their risk profiles per 100 mℓ as stipulated in the WHO (2022) guidelines. Table 8.3 and Table 8.4 shows the classification scales on risk (vulnerability) profiles for total coliforms and *E. coli*, as prescribed by the WHO (2022) guidelines.

Table 8.3 Classification for total coliform colonies per 100 mℓ water sample

Risk level	In conformity	Low risk	Intermediate risk	High risk for potable use
Number of colonies	<1	1–10	11–100	>100

Source: WHO (2022)

Table 8.4 Classification for *E. coli* colonies per 100 mℓ water sample

Risk level	In conformity	Low risk	Intermediate risk	High risk	High risk for potable use
Number of colonies	0	1–10	10–100	100–1 000	>1 000

Source: WHO (2022)

The risk profiles for total coliforms and *E. coli* colonies were compared to the vulnerability indices of each borehole, and the results are shown in Table 8.5 and Table 8.6, respectively.

Table 8.5 Comparison of DWAPH vulnerability index against total coliform concentration

Borehole	Vulnerability index	Vulnerability type (DWAPH classification)	Total Coliform (number per 100 mℓ)	Risk classification. (WHO, 2022)
B1	12	Moderate	2	Low
B2	11	Moderate	> 2 420	Very high
B3	12	Moderate	> 2 420	Very high
B4	10	Low	< 1	Conformity
B5	11	Moderate	127	High risk

Table 8.6 Comparison of DWAPH vulnerability index against *E. coli* concentration

Borehole	Vulnerability index	Vulnerability type (DWAPH classification)	<i>E. coli</i> (number per 100 mℓ)	Risk classification. (WHO, 2022)
B1	12	Moderate	< 1	In conformity
B2	11	Moderate	< 1	In conformity
B3	12	Moderate	< 1	In conformity
B4	10	Low	< 1	In conformity
B5	11	Moderate	< 1	In conformity

The comparison of the vulnerability maps for the DWAPH and total coliform count indicated a minimal correlation between the DWAPH index and data from the total coliform count. Out of the five boreholes, only one borehole (B4) showed a close association between the vulnerability type and risk classification. However, the

comparison between the DWAPH index and *E. coli* count indicated a positive correlation with boreholes B1, B2, B3, and B5, showing a consistent relationship between the moderate scale region and the in-conformity data from the *E. coli* vulnerability chart. Because of this validation, it can be interpreted that the DWAPH index can also be used with consistency in validating other risk indices where scales do not have large ranges or huge variances. The low number of study samples used in this study could have attributed to the poor correlations established between the DWAPH index and the *E. coli* numbers. Figures 8.3 and 8.4 show the risk vulnerability profile maps for the total coliforms and *E. coli* counts against the DWAPH indices.

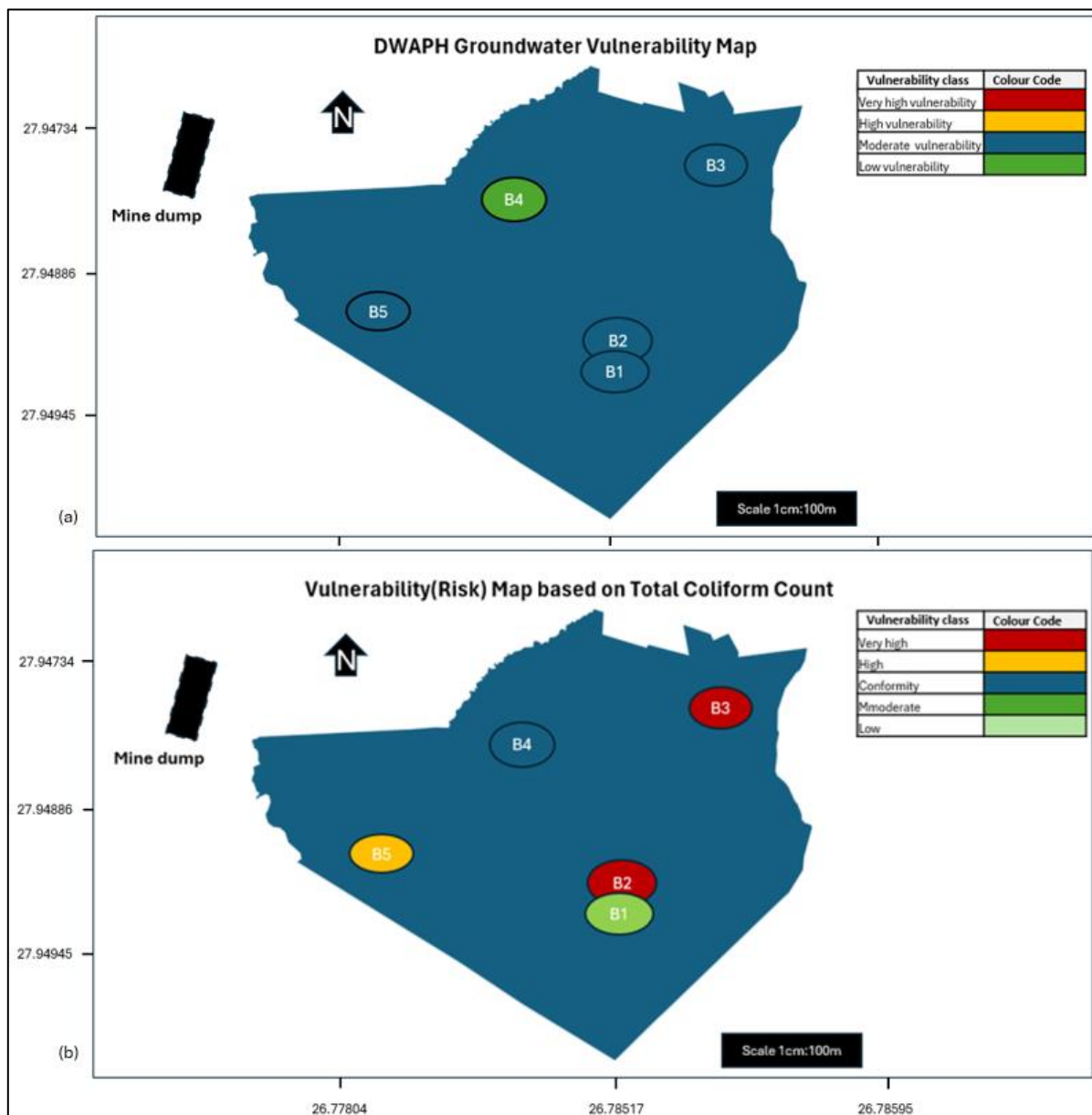


Figure 8.3 Vulnerability maps based on (a) the DWAPH and (b) Total Coliform count for the five boreholes in the study area

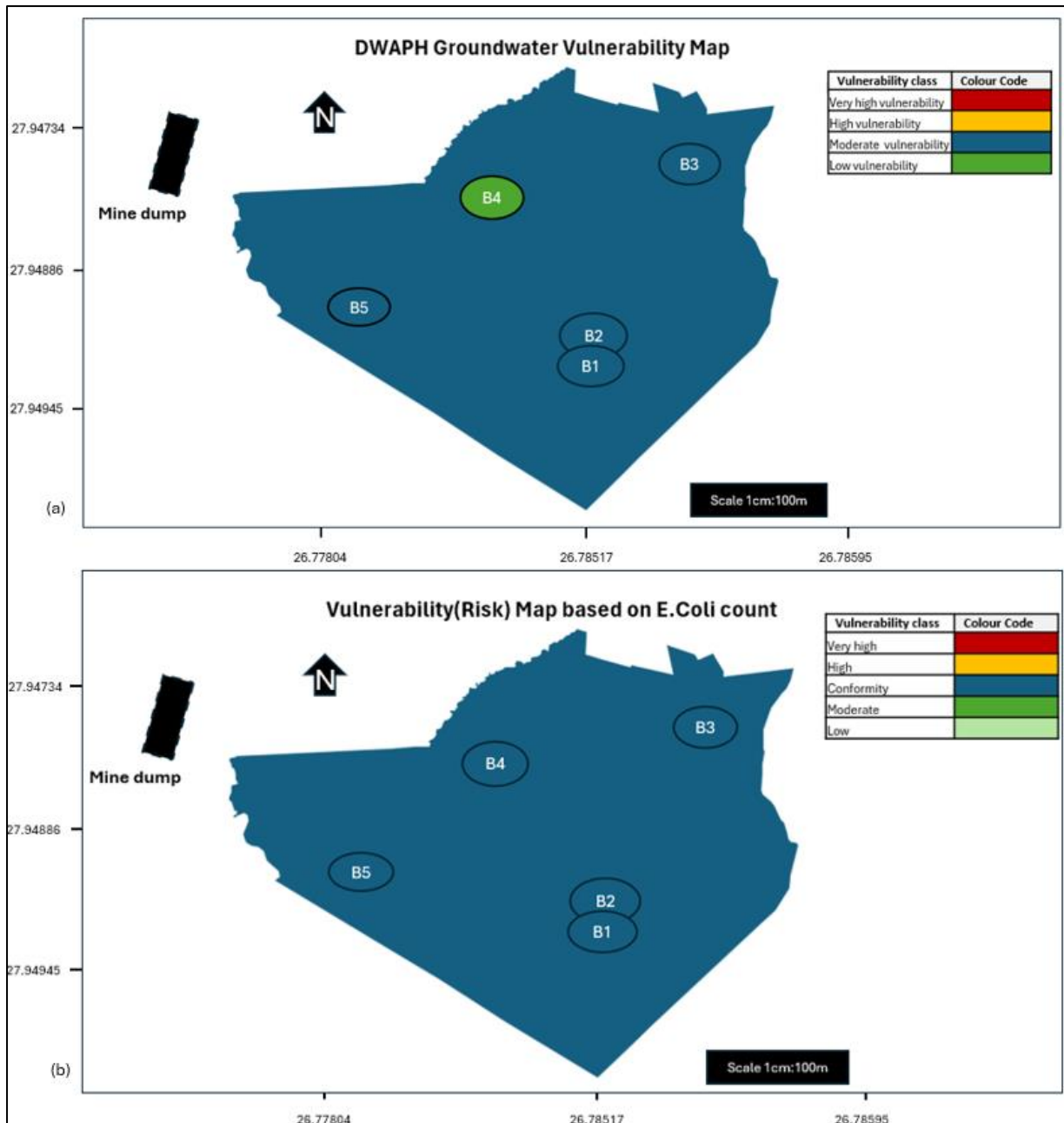


Figure 8.4 Vulnerability maps based on (a) the DWAPH and (b) E. coli count for the five boreholes in the study area

8.4 Significance of study

The development of a specific groundwater vulnerability assessment method is important in tracing and identifying vulnerable groundwater sources from contamination. In this study, the development of the DWAPH method was an important milestone in the tracking and tracing of groundwater vulnerability from mine dumps in South Africa as the method can be replicated in other areas with similar conditions to

that of the Welkom area. The outcomes of this study have implications on the following people, studies, and communities:

8.4.1 Implications on mining communities

Mining communities in South Africa have long been at the forefront of exposure to all different sets of contaminants varying from air, noise, and water pollution (Kamunda et al., 2016). However, without affordable, simple, and reliable methods to detect, measure or assess the extent of vulnerability of communities to the pollutants, are always difficult for the local government authorities to ascertain the extent or source of contamination. The development of the simple vulnerability methods, such as DWAPH, will greatly benefit mining communities in assessing the likelihood of their groundwater sources from mine dump contamination. Furthermore, local authorities will be able to better support, advise, and guide their communities on the utilisation of groundwater.

8.4.2 Implications for groundwater management

It is expected that the outcomes of this study will assist water authorities in better understanding the dynamics of groundwater flow movement in areas next to mine dumps. Unlike other groundwater vulnerability assessment methods, the DWAPH method brings unique parameters such as groundwater flow direction, distance from contaminant source measurements, and water quality which are critical for water managers in mapping groundwater protection and conservation programmes. Furthermore, the DWAPH method will be a valued addition to the already existing groundwater vulnerability assessment methods as it is only one of a few methods that deals with specific vulnerability.

8.4.3 Implications for point source pollution

Despite the existence of several groundwater vulnerability assessments, very few methods have focused on delineating contamination from point sources (Ebenebe et al., 2017; WHO, 2018). The significance of this study is that it provides a method which can be used to assist in assessing the groundwater vulnerability of areas next to mine dumps or other point source pollution sources such as landfills, farms, and graveyards. The design and simplicity of the DWAPH vulnerability assessment method makes it relevant and easy to apply in studies involving point source pollution other than in mine dumps.

8.4.4 Limitations to study and method

The DWAPH method has the potential to be a significant tool in the establishment of groundwater vulnerabilities from point sources. Although the positive validation of the DWAPH method is critical for its acceptance, reliability, and ultimate adoption for use in other studies, it should be noted that every method has its own successes and limitations. Furthermore, the design of the study from which the DWAPH vulnerability assessment was developed also plays a role as it has its own shortfalls. Based on this background, some of the limitations of the study, as well as the DWAPH methodology, are as follows:

- The sampling size of the study was small. The study findings were based on only five boreholes, which could have compromised the significance of the outcomes.
- Data paucity. Other than the conductivity and dilution tests conducted to investigate the aquifer types and aquifer fractures, further tests such as pump tests could have been conducted, including the use of borehole log data.
- More geophysical data should be collected on-site, other than the reliance on literature to obtain study data.
- The inclusion of the direction of flow parameter is very significant in strengthening the DWAPH vulnerability model; however, the number of boreholes from which the groundwater flow direction can be established was very limited.
- The non-inclusion of the weighting in addition to the rating of parameters in the DWAPH method was part of the deliberate design to simplify the method in its application; however, the lack of the weighting may have compromised the efficiency and accuracy of the DWAPH method in detecting groundwater vulnerability.

However, despite these weaknesses and limitations, the DWAPH method can still be considered for application in other areas based on the validations and comparisons with other methods conducted in this study.

8.5 Conclusion

The validation process is an important component of evaluating the functionality and efficiency of groundwater vulnerability assessment models. For this study, the validation

of the DWAPH method with NO_3^- concentrations yielded a weak correlation of $R^2 = 0.0894$. However, like any other validation on groundwater vulnerability assessment methods, large data sample sizes are required in order to clearly establish reliable outcomes. Further validation conducted between the DWAPH index and microbiological data involving total coliforms and *E. coli* counts established weak to strong correlations, respectively. The vulnerability maps between the DWAPH index and the *E. coli* counts showed great correlations in all but one borehole. Based on this validation, it can be concluded that the DWAPH vulnerability method can be reliably applied in cases where data samples are huge, with minimal deviation between variables. However, further validation of the DWAPH method can still be conducted to test the suitable conditions within which the method can be reliably applied.

Chapter 9

Conclusion and Recommendations

9.1 Conclusion

The main aim of this study was to evaluate the vulnerability of the groundwater in Welkom, Free State, South Africa, to contamination from mine dumps. To achieve this, a new groundwater vulnerability assessment method was developed based on the contextual factors that characterises specific vulnerability within the study area. The conclusions of the study are presented under the sub-aims as follows:

9.1.1 Assessing the groundwater vulnerability in the Welkom area to contamination from mine dumps

- A new groundwater vulnerability assessment method was developed based on the hydrological, geohydrological, and hydrogeological characterisations of the study area. Following up to the characterisation, five parameters were subjectively selected based on their ability to detect specific risks associated with the movement of contaminants from the mine dump to the boreholes located at the study site. These five parameters were:

D – Direction to groundwater flow

W – Water quality

A – Aquifer type

P – Precipitation

H – Horizontal distance between contaminant and water source

Each of the five parameters were rated and formed part of the overall DWAPH vulnerability index outcomes that were classified under four classes ranging from low (5–10), to moderate (10–15), to high (15–20), and very high (21–25).

- The DWAPH vulnerability method was applied to assess the groundwater vulnerability in Welkom from a mine dump through the five boreholes located at the CUT campus. The overall vulnerability index score from the assessment was 13, indicating that the vulnerability of the groundwater from the five boreholes was

relatively moderate in relation to the mine dump. Further vulnerability assessments conducted for individual boreholes B1–B5, based on the adjusted distance to contaminant scale, indicated moderate vulnerability scores in all but borehole B4, which had a low vulnerability score.

- The overall water quality index of all the boreholes (B1–B5) was poor (67.8), characterised by high SO_4 and NO_2^- concentrations assumed to be the byproducts from the gold mining process. This was evidence that the groundwater in the Welkom area is contaminated by chemical elements and compounds from the mining process.
- The aquifer type, composed of sandstone, siltstone, and limestone, are characterised as very porous and therefore have an overall contribution towards higher vulnerability of the groundwater to contamination. Additionally, the shorter distances, all less than one kilometre between the boreholes and the mine dump, contributed strongly to greater vulnerability of the groundwater to contamination.

9.1.2 Comparing and validating the effectiveness of the new groundwater vulnerability assessment method to the existing vulnerability assessment methods

- Established groundwater vulnerability assessment methods such as the AVI, GOD, and the RTt methods were applied to assess the groundwater vulnerability to contamination in the study area. The outcomes from the evaluations were as follows:
 - AVI – High vulnerability
 - GOD – Moderate vulnerability
 - RTt – Low vulnerability.

Among the three vulnerability assessment methods tested, only the GOD method produced results that aligned with those of the DWAPH method. This consistency can be attributed to the close alignment of both methods with the specific parameters of the study site. In contrast, the AVI and RTt methods yielded outcomes that diverged from those of the DWAPH method. This discrepancy is likely due to the differing designs of the AVI and RTt methods, which appear to have less relevance to the unique conditions of the study area.

The strong alignment between the outcomes of the GOD and DWAPH methods suggests that the DWAPH method has the potential to provide reliable and credible results when compared to other vulnerability assessment techniques. This consistency highlights the effectiveness of the DWAPH method in capturing site-specific factors, making it a valuable tool for vulnerability analysis in settings similar to the study area. The divergent results from the AVI and RTt methods further underscore the importance of selecting assessment techniques that are well-matched to the conditions being studied.

- On validation with single parameters, the DWAPH method showed a weak correlation with an NO_3^- of $R^2 = 0.0894$. When validated against total coliforms and *E. coli*, the DWAPH method showed a weak to strong relationship, respectively.

9.1.3 Recommendations

The recommendations given in this unit are based on the outcomes of this study. The recommendations are categorised into three sections, which are the safety and usage of groundwater in the Welkom area, potential areas for further studies, and the advice to governments, communities, and the general populace.

9.1.3.1 Recommendations on safety and usage of the groundwater in the Welkom area

Based on the outcomes of this study, the vulnerability of the groundwater from the five boreholes was classified as moderate. This denotes that the groundwater in the study areas has a fair chance of being contaminated by the mine dump. Furthermore, the water quality tests conducted across all five boreholes indicated that the water quality ranged from poor to very poor, thereby showing a high probability of possible contamination. It is therefore recommended that:

- No borehole water at the CUT Welkom campus should be used for drinking purposes as it may pose health risks.
- The groundwater sources close to mine dumps around the Welkom area be continuously retested for water quality as well as against the DWAPH vulnerability index to determine the risk posed by mine dumps.

- Remediation methods such as the pump and treat system should be applied to treat the water and ensure that it is safe for human and animal consumption. In the case of accidental ingestion or suspected poisoning from the groundwater, students and staff at the CUT campus should seek urgent medical attention.
- Water from the effluent stream surrounding the mine dump should be closely monitored as it poses a serious health risk to children and animals. The water quality tests conducted indicated that the water had high levels of ions such as Ca, Mg, SO₄, NO₃, and TDS, all above the SANS 241 drinking water quality standards.

9.1.3.2 Recommendations for further studies

The study developed the groundwater vulnerability assessment method for specifically assessing the vulnerability posed by mine dumps to groundwater sources. However, due to the nature of the research, several limitations were encountered but could not be addressed due to resource limitations. These limitations can, however, be remediated in further studies. The recommendations for further studies are outlined as follows:

- Further studies should utilise larger sample sizes of water sources to improve the quality and statistical significance of the outcomes.
- The performance of the DWAPH method needs to be applied and assessed in other aquifer types such as confined and Karst landforms to establish the limitations of the method.
- The DWAPH method should be conducted in different areas to ascertain its reliability and accuracy.
- The geological characterisation of the study area should involve more field tests rather than depend on literature, as field tests give more context and relevancy to the outcomes.
- The DWAPH vulnerability assessment method should be compared to more vulnerability assessment methods, including the modified assessments to establish its accuracy relative to other methods.

- Other than the NO₃⁻ and coliform validation conducted in this study more single parameter analyses should be used to validate the outcomes from the DWAPH method.
- The further improvement of the DWAPH method can be conducted by adding weights to the ratings of the parameters where applicable.

9.1.3.3 Recommendations to governments, communities, and the general populace

The purpose of this study was to assess the impact of the proximity of the mine dump to groundwater sources. Despite the moderate vulnerability identified by the DWAPH assessment, the water quality of the boreholes calculated across all four seasons indicated a high possibility of contamination of the groundwater from the mine dump. The following recommendations apply to the governments, communities, and the general populace with regard to the future of groundwater use in the Welkom area:

- The local government should run awareness programmes to inform the public on the dangers of drinking groundwater in the area as there is a possibility of contamination by mine dump elements.
- Local governments should improve on the quality and efficiency of their water distribution system to encourage the use of tap water and minimise the reliance of groundwater.
- The local and national governments should engage and monitor mining companies to participate in the restoration of mining waste to minimise the leakage and contamination of groundwater.
- The mine authorities should be advised to better manage the effluent from the mine to reduce the high number of contaminants recorded in the effluent stream which may pose serious health risks to the environment.
- The national government should enforce regulations for the management of abandoned mine dumps to guard against the contamination of groundwater sources.

- The students and staff at the CUT Welkom Campus should not use the groundwater in any of the five boreholes for drinking purposes, nor should it be given to animals.
- Mine dumps in the Welkom area should be better designed on the basis of the hydrological data and information to protect the groundwater sources.

References

Abbasi, S., Mohammadi, K., Kholghi, M. and Howard, K. (2013). Aquifer vulnerability using DRASTIC, weights of evidence and the analytic element method = Evaluation de la vulnérabilité des aquifères à l'aide de DRASTIC, de la méthode de pondération des preuves et de la méthode des éléments analytiques. *Hydrological Sciences Journal*, 58(1):186-197. <https://doi.org/10.1080/02626667.2012.743027>

Abdalla, O.A.E., Suliman, F.O., Al-Ajmi, H., Al-Hosni, T. and Rollinson, H. (2010). Cyanide from gold mining and its effect on groundwater in arid areas, Yanqul mine of Oman. *Environmental Earth Sciences*, 60(4):885-892. <https://doi.org/10.1007/s12665-009-0225-z>

Abdullah, T.O. (2020). Groundwater vulnerability assessment to pollution in different soil and rock materials. Division of Mining and Geotechnical Engineering. Doctoral thesis, Environmental and Natural Resources Engineering. Luleå University of Technology, Sweden. <http://www.diva-portal.org/smash/get/diva2:1439884/FULLTEXT01.pdf>

Abegunde, O.A. (2014). Geologic and geochemical assessment of acid mine drainage and heavy metals contamination in the West Rand, Witwatersrand Basin, South Africa. Masters dissertation. Faculty of Natural Sciences, University of the Western Cape. <http://hdl.handle.net/11394/4785>

Abraham, J., Dowling, K. and Florentine, S. (2018). Assessment of potentially toxic metal contamination in the soils of a legacy mine site in Central Victoria, Australia. *Chemosphere* 192:122-132. <https://doi.org/10.1016/j.chemosphere.2017.10.150>

Adetunji, V.O. and Odetokun, I. (2011). Groundwater contamination in Agbowo community, Ibadan Nigeria: Impact of septic tanks distances to wells. *Malaysian Journal of Microbiology*, 7(3):159-166. <http://dx.doi.org/10.21161/mjm.33011>

Ait Bahammou, Y., Benamara, A., Ammar, A. and Dakir, I. (2019). Fracture zones detection for groundwater exploration integrating resistivity profiling and very low

frequency electromagnetic methods (Errachidia basin, Morocco). *Contributions to Geophysics and Geodesy*, 49(2):181-194. <https://doi.org/10.2478/congeo-2019-0009>

Akaahan, T.J., Oluma, H.O.A. and ShaíAto, R. (2010). Physico-chemical and bacteriological quality of water from shallow wells in two rural communities in Benue State, Nigeria. *Pakistan Journal of Analytical and Environmental Chemistry*, 11(1):73-78. <https://www.researchgate.net/publication/267817549>

Akinlalu, A.A., Mogaji, K.A. & Adebodun, T.S. (2021). Assessment of aquifer vulnerability using a developed “GODL” method (modified GOD model) in a schist belt environ, Southwestern Nigeria. *Environmental Monitoring and Assessment*, 193(199). <https://doi.org/10.1007/s10661-021-08960-z>

Albinet, M. and Margat, J. (1970). Cartographie de la vulnérabilité à la pollution des nappes deau souterraine Orleans, France. *Bull BRGM, 2ème série*, 4:13-22.

Ali, S.S. (2007) Geology and hydrogeology of Sharazoor-Piramagroon Basin in Sulaimani area, northeastern Iraq. Doctoral thesis, Faculty of Mining and Geology, University of Belgrade, Belgrade, Serbia.

Al Kuisi, M., El-Naqa, A. and Hammouri, N. (2006). Vulnerability mapping of shallow groundwater aquifer using SINTACS model in the Jordan Valley area, Jordan. *Environmental Geology*, 50(5):651-667. <https://doi.org/10.1007/s00254-006-0239-8>

Aller, L., Bennet, T., Hehr, J.H., Petty, R.J. and Hackett, G. (1987). DRASTIC: A standardized system for evaluating groundwater pollution potential using hydrogeologic settings. U.S. Environmental Protection Agency.

Anantha Rao, D., Naik, P.K., Jain, S.K., Vinod Kumar, K. and Dhanamjaya Rao, E.N. (2018). Assessment of vulnerability zones for ground water pollution using GIS-DRASTIC-EC model: A field-based approach. *Journal of Earth System Science*, 127, article 49. <https://doi.org/10.1007/s12040-018-0944-1>

Ashbolt, N.J., Grabow, W.O.K. and Snozzi, M. (2001). Indicators of microbial water quality. In: Fewtrell, L. & Bartram, J. (Eds.), *Water quality: Guidelines, standards and health assessment of risk and risk management for water-related infectious disease*.

Geneva: World Health Organization (pp. 289-315). World Health Organization.
<https://iris.who.int/bitstream/handle/10665/42442/924154533X.pdf>

ASTM International. D6001 – 05. (2012). *Standard guide for direct-push groundwater sampling for environmental site characterization*. 100 Barr Harbor Dr., W. Conshohocken, PA 19428.

Ayonkule, A.O. (2014). *Geologic And Geochemical Assessment of Acid Mine Drainage and Heavy Metals Contamination in The West Rand, Witwatersrand Basin, South Africa*. Master's dissertation, Faculty of Natural Sciences, University of the Western Cape.
<http://hdl.handle.net/11394/4785>

Avseth, P., Flesche, H. and Van Wijngaarden, A-J. (2003). AVO classification of lithology and pore fluids constrained by rock physics depth trends. *The Leading Edge*, 22(10):921-1056. <https://library.seg.org/doi/reader/10.1190/1.1623641>

Balasubramanian, A. (2017). *Procedure for conducting pumping tests*.
<https://doi.org/10.13140/RG.2.2.18948.32641>

Barbulescu, A. (2020). Assessing groundwater vulnerability: DRASTIC and DRASTIC-like methods: A review. *Water* 2020,12(5), article 1356. <https://doi.org/10.3390/w12051356>

Benson, R.C. and Yuhr, L.B. (2016). *Site characterization in Karst and pseudokarst terraines: Practical strategies and technology for practicing engineers, hydrologists and geologists*. Dordrecht, Netherlands: Springer. https://doi.org/10.1007/978-94-017-9924-9_11

Beukes, N. and Nelson, J. (1995). Sea-level fluctuation and basin subsidence controls on the setting of auriferous palaeoplacers in the Archaean Witwatersrand Supergroup: A genetic and sequence stratigraphic approach. *Centennial Geocongress, Extended Abstracts*, pp. 860-863.

Belle, G.N. (2021). *Contamination of water, soil, and plants by contaminants from gold mine tailings in Matjhabeng Local Municipality, South Africa*. Doctoral thesis, Central University of Technology, Free State. <http://hdl.handle.net/11462/2335>

Belle, G., Fossey, A., Esterhuizen, L. and Moodley, R. (2021). Contamination of groundwater by potential harmful elements from gold mine tailings and the implications to human health: A case study in Welkom and Virginia, Free State Province, South Africa. *Groundwater or Sustainable Development*, 12, article 100507.

<https://doi.org/10.1016/j.gsd.2020.100507>

Benacchio, S. and Civita, M. (1988). Difesa degli acquiferi dell' alta pianura veneta, stato di inquinamento e vulnerabilita delle acque sotteranee del bacino del brenta. Monography G.N.D.C.I.–C.N:R., vol 5, Venice.

Boggs, S. (2009). *Petrology of Sedimentary Rocks*. Cambridge University Press.

Boggs, S. Jr. (1995). *Principles of sedimentology and stratigraphy*. Prentice Hall, New Jersey.

Bogomolov, G.V. (1966). Fissure and karst waters. *International Association of Scientific Hydrology. Bulletin*, 11(4):13-17. <https://doi.org/10.1080/02626666609493490>

Bokar, H., Traoré, A.Z., Mariko, A., Diallo, T., Traoré, A., Sy, A., Soumaré, O., Dolo, A., Bamba, F., Sacko, M. and Touré, O. (2020). Geogenic influence and impact of mining activities on water soil and plants in surrounding areas of Morila Mine, Mali. *Journal of Geochemical Exploration*, 209, article 106429.

<https://doi.org/10.1016/j.gexplo.2019.106429>

Bonton, A., Bouchard, C., Rouleau, A., Rodriguez, M.J. and Therrien, R. (2012). Calibration and validation of an integrated nitrate transport model within a well capture zone. *Journal of Contaminant Hydrology*, 128(1-4):1-18,

<https://doi.org/10.1016/j.jconhyd.2011.10.007>

Brahmi, S., Baali, F., Hadji, R., Brahmi, S., Hamad, A., Rahal, O., Zerrouki, H., Saadali, B. and Hamed, Y. (2021). Assessment of groundwater and soil pollution by leachate using electrical resistivity and induced polarization imaging survey, case of Tebessa municipal landfill, NE Algeria. *Arabian Journal of Geosciences*, 14, article 249.

<https://doi.org/10.1007/s12517-021-06571-z>

Breiman, L. (2001). Random forests. *Machine Learning*, 45(1):5-32.

<https://doi.org/10.1023/A:1010933404324>

Brown, R.M., McClelland, N.I., Deininger, R.A. and Tozer, R.G. (1970). A water quality index – Do we dare. *Water and Sewage Works*, 117(10):339-343.

Brown, R.M., McClelland, N.I., Deininger, R.A., O'Connor, M.F. (1973). Water quality index – Crashing the psychological barrier. In Jenkins, S.H. (Ed.), *Advances in water pollution research*, Proceedings of the Sixth International Conference held in Jerusalem, June 18-23, 1972, pp. 787-797. Oxford: Pergamon Press. <https://doi.org/10.1016/B978-0-08-017005-3.50067-0>

Bumby, A.J., and Eriksson, P.G. (2021). *Sedimentology and Basin Evolution*.

Burke, J and Villholth, K. (2007). Groundwater: A global assessment of scale and significance, Chapter 10, pp. 396-423. International Water Management Institute. <https://www.iwmi.cgiar.org/assessment/Water%20for%20Food%20Water%20for%20Life/Chapters/Chapter%2010%20Groundwater.pdf>

Camejo, J., Pacheco, O.R. and Guevara, M. (2013). Study of water quality classifiers. In: Proceedings of the Transboundary water management across borders and interfaces conference. TWAM2013 International Conference & Workshops. Aveiro, Portugal.

CCME (Canadian Council of Ministers of the Environment). 2001. *Canadian water quality guidelines for the protection of aquatic life: CCME Water Quality Index 1.0, User's Manual*. In Canadian environmental quality guidelines, 1999. Winnipeg.

Chakraborty, S. and Kumar, R.N. (2016). Assessment of groundwater quality at a MSW landfill site using standard and AHP based water quality index: A case study from Ranchi, Jharkhand, India. *Environmental Monitoring and Assessment*, 188, article 335. <https://doi.org/10.1007/s10661-016-5336-x>

Civita, M.V. (2010). The combined approach when assessing and mapping groundwater vulnerability to contamination. *Journal of Water Resource and Protection*, 2(1):14-28. <https://doi:10.4236/jwarp.2010.21003>

Civita, M. and De Maio, M. (1997). SINTACS. Un sistema parametrico per la valutazionee la cartografia della vulnerabilita` degli acquiferi all'inquinamento. *Metodologia and Automatizzazione*, vol. 60. Pitagora Editrice, Bologna, Italia.

Civita, M. and De Maio, M. (2000). SINTACS R5, a new parametric system for the assessment and automatic mapping of ground water vulnerability to contamination. Pitagora Editrice, Bologna Climate data.

Climate Data. (2024). Welkom Climate (South Africa). *Data and graphs for weather & climate in Welkom*. Climate Data. <https://en.climate-data.org/africa/south-africa/free-state/welkom-765253/> [Accessed 02 January 2024]

Cook G. P. (2003). *A guideline to regional groundwater in fractured rock aquifers*. National Library of Australia Cataloguing. CSIRO Land and Water, Glen Osmond, SA, Australia

Cossu, R., Zuffianò, L.E., Limoni, P.P., De Giorgio, G., Pizzardini, P., Miano, T., Mondelli, D., Garavaglia, R., Carella, C. and Polemio, M. (2018). How can the role of leachate on nitrate concentration and groundwater quality be clarified? An approach for landfills in operation (Southern Italy). *Waste Management*, 77:156-165.

<https://doi.org/10.1016/j.wasman.2018.05.014>

Crane, S.R. and Moore, J.A. (1984). Bacterial pollution of groundwater: A review. *Water, Air, and Soil Pollution*, 22:67-83. <https://doi.org/10.1007/BF00587465>

Dakir, I., Benamara, A., Aassoumi, H., Ouallali, A. and Bahammou, Y.A. (2019). Application of Induced polarization and resistivity to the determination of the location of metalliferous veins in the Taroucht and Tabesbaste areas (Eastern Anti-Atlas, Morocco). *International Journal of Geophysics*, article 5849019.

<https://doi.org/10.1155/2019/5849019>

Daly, D., Dassargues, A., Drew, D., Dunne, S., Goldscheider, N., Neale, S., Popescu, I. and Zwahlen, F. (2002). Main concepts of the European approach for (karst) groundwater vulnerability assessment and mapping. *Hydrogeology Journal*, 10:340-345.

<https://doi.org/10.1007/s10040-001-0185-1>

Dankert, B.T. and Hein, K.A.A. (2010). Evaluating the structural character and tectonic history of the Witwatersrand basin. *Precambrian Research*, 177:1-22.

<https://doi.org/10.1016/j.precamres.2009.10.007>

DWAF (Department of Water Affairs and Forestry). (2000). Policy and strategy for groundwater quality management in South Africa. Number W.1.0: 1st Ed.

Ding, L., Yang, Q., Yang, Y., Ma, H. and Martin, J.D. (2020). Potential risk assessment of groundwater to address the agricultural and domestic challenges in Ordos Basin. *Environmental Geochemistry and Health*, 43:717-732. <https://doi.org/10.1007/s10653-019-00512-2>

Doerfliger, N. and Zwahlen, F. (1998). *Groundwater vulnerability mapping in karstic regions (EPIK). Practical guide: Application to groundwater protection zones*. Swiss Agency for the Environment, Forests and Landscape, Bern.

Dong, S., Zeng, L., Lyu, W., Xia, D., Liu, G., Wu, Y and Du, X. (2020). Fracture identification and evaluation using conventional logs in tight sandstones: A case study in the Ordos Basin, China. *Energy Geoscience*, 1(3-4):115-123.

<https://doi.org/10.1016/j.engeos.2020.06.003>

Driscoll, C.T., Mason, R.P., Chan, H.M., Jacob, D.J. and Pirrone, N. (2013). Mercury as a global pollutant: Sources, pathways, and effects. *Environmental Science and Technology*, 2013, 47(10):4967-4983. <https://doi.org/10.1021/es305071v>

Ducci, D. and Sellerino, M. (2022). A modified AVI model for groundwater vulnerability mapping: Case studies in Southern Italy. *Water (Switzerland)*, 14(2), article 248. <https://doi.org/10.3390/w14020248>

Durand, J.F. (2012). The impact of gold mining on the Witwatersrand on the rivers and karst system of Gauteng and North West Province, South Africa. *Journal of African Earth Sciences*, 68:24-43. <https://doi.org/10.1016/j.jafrearsci.2012.03.013>

DWAF (Department of Water Affairs and Forestry). (2000). *Policy and strategy for groundwater quality management in South Africa*. Number W.1.0. Pretoria.

DWAF (Department of Water Affairs and Forestry). (2004a). *Groundwater Protection. Guidelines for protecting boreholes and wells. Toolkit for water services*. Number 3.3. Pretoria.

DWAF (Department of Water Affairs and Forestry). (2004b). *Guidelines on protecting groundwater from contamination. Toolkit for water services*. Number 3.4. Pretoria.

Dzwaitiro, B., Hoko, Z., Love, D. and Guzha, E. (2006). Assessment of the impacts of pit latrines on groundwater quality in rural areas: A case study from Marondera district, Zimbabwe. *Physics and Chemistry of the Earth, Parts A/B/C*, 31(15-16):779-788. <https://doi.org/10.1016/j.pce.2006.08.031>

Ebenebe, P.N. (2016). Evaluation of trace element in the environment around the Lejweleputswa area of the Free State, South Africa. Doctoral thesis, Department of Environmental Sciences, Central University of Technology.

<http://hdl.handle.net/11462/2322>

Ebenebe, P.C., Shale, K., Sedibe, M.M., Tikilili, P.V. and Achilonu, M.C. (2017). South African mine effluents: Heavy metal pollution and impact on the ecosystem. *International Journal of Chemical Sciences*, 15(4), article 198.

Eriksson, P.G., Altermann, W. and Hartzler, F.J. 2006. The Transvaal supergroup and its precursors. In: Johnson, M.R., Anhaeusser, C.R. and Thomas, R.J. (Eds.), *The Geology of South Africa* (pp. 237-260). Geological Society of South Africa and the Council for Geoscience.

Environmental Protection Agency. (2003). National management measures to control nonpoint source pollution from agriculture. US EPA 841-B-03-004. <https://www.epa.gov/nps/national-management-measures-control-nonpoint-source-pollution-agriculture>

Ewusi, A., Ahenkorah, I. and Kuma, J.S.Y. (2017). Groundwater vulnerability assessment of the Tarkwa mining area using SINTACS approach and GIS. *Ghana Mining Journal*, 17(1):18-30. <https://doi.org/10.4314/gm.v17i1.3>

Falowo, O.O., Akindureni, Y. and Ojo, O. (2017). Groundwater assessment and its intrinsic vulnerability studies using aquifer vulnerability index and GOD methods. *International Journal of Energy and Environmental Science*, 2(5):103-116. <http://dx.doi.org/10.11648/j.ijees.20170205.13>

Fashola, M.O., Ngole-Jeme, V.M., & Babalola, O.O. (2016). Heavy metal pollution from gold mines: Environmental effects and bacterial strategies for resistance. *International Journal of Environmental Research and Public Health*, 13(11), article 1047. <https://doi.org/10.3390%2Fijerph13111047>

Focazio, M.J., Szabo, Z., Kraemer, T.F., Mullin, A.H., Barringer, T.H. and DePaul, V.T. (2003). *Occurrence of selected radionuclides in ground water used for drinking water in the United States: A reconnaissance survey, 1998*. Water-Resources Investigations Report 00-4273. U.S. Geological Survey.

<https://pubs.usgs.gov/wri/wri004273/pdf/wri004273.pdf>

Foster, S.S.D. (1987). Fundamental concepts in aquifer vulnerability, pollution risk and protection strategy. International Conference, 1987, Noordwijk Aan Zee, the Netherlands. In: Van Duijvenbooden, W. and Van Waegeningh, H.G. (Eds.), *Vulnerability of soil and groundwater to pollutants*. The Hague: Committee for Hydrological Research TNO.

Frimmel, H.E. and Minter, W.E.L. (2002). Recent developments concerning the geological history and genesis of the Witwatersrand gold deposits, South Africa. In: Goldfarb, R.J. and Nielsen, R.L. (Eds.), *Integrated methods for discovery: Global exploration in the twenty-first century*, Vol. 9. Society of Economic Geologists. <https://doi.org/10.5382/SP.09.02>

Frimmel, H.E., Groves, D.I., Kirk, J., Ruiz, J., Chesley, J., Minter, W.E.L. (2005). The formation and preservation of the Witwatersrand goldfields, the world's largest gold Province. In: Hedenquist, J.W., Thompson, J.F.H., Goldfarb, R.J. and Richards, J.P. (Eds.), *One hundredth anniversary volume* (pp. 769-797). Littleton Co.: Society of Economic Geologists. <https://doi.org/10.5382/AV100.23>

Google Maps. (2023). https://www.google.com/maps/@-27.9484942,26.7809802,16.25z/data=!4m3!1m2!2sv_Qv9I4yRSi57QBv7oNNBg!3e3?entry=ttu. [Accessed 02 June 2023]

Gorde, S.P. and Jadhav, M.V. (2013). Assessment of water quality parameters: A review. *Journal of Engineering Research and Applications*, 3(6):2029-2035. https://www.ijera.com/papers/Vol3_issue6/LV3620292035.pdf

Gowrisankar, G., Chelliah, R., Ramakrishnan, S.R., Elumalai, V., Dhanamadhavan, S., Brindha, K., Usha, A. and Elango, L. (2017). Chemical, microbial and antibiotic susceptibility analyses of groundwater after a major flood event in Chennai. *Scientific Data*, 4(1), article 170135. <https://doi.org/10.1038/sdata.2017.135>

Greater Oliver Groundwater Vulnerability. (n.d.) *Introduction to DRASTIC*. Cyrille M'dard de Chardon. https://www.sfu.ca/personal/dallen/Visualization/DRASTIC/drastic_page.html (Accessed: 17 January 2024).

Green Door Environmental. (2013). Proposed establishment of a bioenergy facility and related infrastructure at Harmony Gold Mine in Welkom, Matjhabeng Local Municipality, Free State Province. Draft scoping report and application: Prepared for Harmony Gold Mining Company Limited, 28 February 2013. <http://www.sahra.org.za/sahris/sites/default/files/additionaldocs/Draft%20Scoping%20Report%2028%20Feb%202013.pdf>

Guo, L., Chehata, N., Mallet, C. and Boukir, S. (2011). Relevance of airborne lidar and multispectral image data for urban scene classification using random forests. *ISPRS Journal of Photogrammetry and Remote Sensing*, 66(1):56-66. <https://doi.org/10.1016/j.isprsjprs.2010.08.007>

Gutzmer, J., and Beukes, N.J. (2020). *Geology and Genesis of Major Copper and Iron Deposits*.

Haith, D.A. and Laden, EM. (1989). Screening of groundwater contaminants by travel-time distributions. *Journal of Environmental Engineering*, 115(3).

[https://doi.org/10.1061/\(ASCE\)0733-9372\(1989\)115:3\(497\)](https://doi.org/10.1061/(ASCE)0733-9372(1989)115:3(497))

Hansen, R.N. (2015). Contaminant leaching from gold mining tailings dams in the Witwatersrand Basin, South Africa: A new geochemical modelling approach. *Applied Geochemistry*, 61:217-223. <https://doi.org/10.1016/j.apgeochem.2015.06.001>

Hao, Q. and Jiang, C. (2015). Heavy metal concentrations in soils and plants in Rongxi manganese mine of Chongqing, southwest of China. *Acta Ecologica Sinica*, 35(1):46-51. <https://doi.org/10.1016/j.chnaes.2015.01.002>

Harter, T. (2008). Assessing vulnerability of groundwater. In: Harter, T. and Rollins, L. (Eds.), *Watersheds, groundwater and drinking water: A practical guide* (pp. 183-204). University of California, Division of Agricultural and Natural Resources.

Harter, T. and Rollins, L. (Eds.) (2008). *Watersheds, groundwater and drinking water: A practical guide*. University of California, Division of Agricultural and Natural Resources.

Hartmann, T. and Jüpner, R. (2020). Implementing resilience in flood risk management. *Wiley Interdisciplinary Reviews: Water*, 7(6), article e1465.

<https://doi.org/10.1002/wat2.1465>

Horton, R.K. (1965). An index number system for rating water quality. *Journal Water Pollution Control Federation*, 37(3):300-306.

Huan, H., Wang, J. and Teng, Y. (2012). Assessment, and validation of groundwater vulnerability to nitrate based on a modified DRASTIC model: A case study in Jilin City of northeast China. *Science of the Total Environment*, 440:14-23.

<https://doi.org/10.1016/j.scitotenv.2012.08.037>

Hussain, Y., Ullah, S.F., Dilawar, A., Akhter, G., Martinez-Carvajal, H. and Roig, H.L. (2022). Assessment of the pollution potential of an aquifer from surface contaminants in a geographic information system: A case study of Pakistan. *Geo-Chicago*, 269, 623–632.

<https://doi.org/10.1061/9780784480120.063>

Isikwue, M.O., Iorver, D. and Onoja, S.B. (2011). Effect of depth on microbial pollution of shallow wells in Makurdi Metropolis [sic], Benue State, Nigeria. *International Journal of Environment and Climate Change*, 1(3):66-73. <https://doi.org/10.9734/BJECC/2011/354>

Javadi, S., Kavehkar, N., Mousavizadeh, M.H. and Mohammadi, K. (2011). Modification of DRASTIC model to map groundwater vulnerability to pollution using nitrate measurements in agricultural areas. In *Journal of Agriculture Science and Technology*, 13:239-249. <https://jast.modares.ac.ir/article-23-10411-en.pdf>

Jhariya, D.C., Khan, R. and Thakur, G.S. (2016). Impact of mining activity on water resource: An overview study. *National Seminar on Recent Practices and Innovations in Mining Industry* at Raipur, pp. 271-277.

John, L. (2012). *Secondary cities in South Africa. The start of a conversation*. Background Report. South African Cities Network. https://www.sacities.net/wp-content/uploads/2020/03/secondary_cities_in_south_africa.2012-2.pdf

Johnson, M.R and Wolmarans, L.G. (2008). *Geology map of South Africa*. Council for Geoscience. <https://maps.geoscience.org.za/portal/apps/sites/#/council-for-geoscience-interactive-web-map1/apps/f17eb56408b34add9c43878de5ccd4c2/explore> [Accessed 07 April 2022]

Kamunda, C., Mathuthu, M. and Madhuku, M. (2016). Health risk assessment of heavy metals in soils from Witwatersrand gold mining basin, South Africa. *International Journal of Environmental Research and Public Health*, 13(7), article 663. <https://doi.org/10.3390/ijerph13070663>

Kanda, A., Ncube, F., Makumbe, P. (2021). Trace elements in groundwater near an abandoned mine tailings dam and health risk assessment (NE Zimbabwe). *Water SA*, 47(4):446-455. <https://doi.org/10.17159/wsa/2021.v47.i4.3851>

Khosravi, K., Sartaj, M., Tsai, F.T.C., Singh, V.P., Kazakis, N., Melesse, A.M., Prakash, I., Tien Bui, D. and Pham, B.T. (2018). A comparison study of DRASTIC methods with various objective methods for groundwater vulnerability assessment. *Science of the Total Environment*, 642, 1032-1049. <https://doi.org/10.1016/j.scitotenv.2018.06.130>

Kinna, R. (2016). Non-discrimination and liability for transboundary acid mine drainage pollution of South Africa's rivers: Could the UN Watercourses Convention open Pandora's mine? *Water International*, 41(3):371-391.

<https://doi.org/10.1080/02508060.2016.1153302>

Kirsch, R. (2006). *Groundwater Geophysics - A Tool for Hydrogeology*. Springer-Verlag, Berlin Heidelberg, 548 pp. <https://doi.org/10.1007/978-3-540-88405-7>

Kogovšek, J. and Petrič, M. (2013). Increase of vulnerability of karst aquifers due to leakage from landfills. *Environmental Earth Sciences*, 70(2):901-912.

<https://doi.org/10.1007/s12665-012-2180-3>

Kossoff, D., Dubbin, W.E., Alfredsson, M., Edwards, S.J., Macklin, M.G. and Hudson-Edwards, K.A. (2014). Mine tailings dams: Characteristics, failure, environmental impacts, and remediation. *Applied Geochemistry*, 51:229-245.

<https://doi.org/10.1016/j.apgeochem.2014.09.010>

Kotchoni, D.O.V., Vouillamoz, J.M., Lawson, F.M.A., Adjomayi, P., Boukari, M. and Taylor, R.G. (2019). Relationships between rainfall and groundwater recharge in seasonally humid Benin: A comparative analysis of long-term hydrographs in sedimentary and crystalline aquifers. *Hydrogeology Journal*, 27:447-457. <https://doi.org/10.1007/s10040-018-1806-2>

Kroeze, C., Gabbert, S., Hofstra, N., Koelmans, A.A., Li, A., Löhr, A., Ludwig, F., Strokal, M., Verburg, C., Vermeulen, L., Van Vliet, T.H., De Vries, W., Wang, M. and Van Wijnen, J. (2016). Global modelling of surface water quality: A multi-pollutant approach. *Current Opinion in Environmental Sustainability*, 23:35-45.

<https://doi.org/10.1016/j.cosust.2016.11.014>

Krumbein, W.C. and Sloss, L.C. (1963). *Stratigraphy and sedimentation*. San Francisco: WH Freeman and Company.

Kumar, S., Thirumalaivasan, D., Radhakrishnan, N. and Mathew, S. (2013). Groundwater vulnerability assessment using SINTACS model. *Geomatics, Natural Hazards and Risk*, 4(4):339-354. <https://doi.org/10.1080/19475705.2012.732119>

Kurwadkar, S. (2017). Groundwater pollution and vulnerability assessment. *Water Environment Research*, 89(10):1561-1577.

<https://doi.org/10.2175/106143017x15023776270584>

Kwon, E., Park, J., Park, W-B., Kang, B.R., Hyeon, B-S. and Woo, N.C. (2022). Nitrate vulnerability of groundwater in Jeju Volcanic Island, Korea. *Science of the Total Environment*, 807(2), article 151399. <https://doi.org/10.1016/j.scitotenv.2021.151399>

Leonard, R.A., Knisel, W.G. and Still, D.A. (1987). GLEAMS: Groundwater loading effects of agricultural management systems. *Transactions of the ASAE*, 30(5):1403-1418. <https://doi.org/10.13031/2013.30578>

Li, N., Han, Z., Ren, Y., Wang, S., Hu, X. and Zeng, Z. (2022). Retarding performance of the vadose zone for nitrogen pollutants derived from municipal solid waste landfills in the red bed zone. *Journal of Environmental Management*, 306, article 114406.

<https://doi.org/10.1016/j.jenvman.2021.114406>

Li, P., Karunanidhi, D., Subramani, T., Srinivasamoorthy, K. (2021). Sources and consequences of groundwater contamination. *Archives of Environmental Contamination and Toxicology*, 80:1-10. <https://doi.org/10.1007/s00244-020-00805-z>

Liang, Y., Yi, X., Dang, Z., Wang, Q., Luo, H. and Tang, J. (2017). Heavy metal contamination and health risk assessment in the vicinity of a tailing pond in Guangdong, China. *International Journal of Environmental Research and Public Health*, 14(12), article 1557. <https://doi.org/10.3390/ijerph14121557>

Liang, J., Li, Z., Yang, Q., Lei, X., Kang, A. and Li, S. (2019). Specific vulnerability assessment of nitrate in shallow groundwater with an improved DRSTIC-LE model. *Ecotoxicology and Environmental Safety*, 174:649-657.

<https://doi.org/10.1016/j.ecoenv.2019.03.024>

Lindström, R. (2005). Groundwater vulnerability assessment using process-based models. Doctoral thesis, KTH Architecture and the Built Environment. <https://www.diva-portal.org/smash/get/diva2:8001/FULLTEXT01.pdf>

Linggasari, S., Cahyadi, T.A., Ernawati, R. and Utami, A. (2020). Analysis of groundwater vulnerability in coal mining at Tanah Laut District, South Kalimantan. *Geographia Technica*, 15(Special Issue):64-73. http://dx.doi.org/10.21163/GT_2020.151.23

Lobo-Ferreira, J.P. and Oliveira, M.M. (2003). *DRASTIC groundwater vulnerability mapping of Portugal*. Proceedings from the 27th Congress of the International Association for Hydraulic Research, San Francisco, USA, Aug. 10-15, pp. 132-137. <https://www.iahr.org/library/infor?pid=25831>

Lottermoser, B.J. (2010). *Mine wastes: Characterization, treatment and environmental impacts*. (Third ed.) Springer. <https://doi.org/10.1007/978-3-642-12419-8>

Love, A.J., Cook, P.G., Harrington, G.A. and Simmons, C.T. (2002). *Groundwater flow in the Clare Valley*. Department for Water Resources, South Australia. Special report DWR02.03.0002.

Lumb, A., Sharma, T.C. and Bibeault, J-F. (2011). A review of genesis and evolution of water quality index (WQI) and some future directions. *Water Quality, Exposure and Health*, 3(1):11-24. <https://doi.org/10.1007/s12403-011-0040-0>

Machiwal, D., Jha, M.K., Singh, V.P. and Mohan, C. (2018). Assessment and mapping of groundwater vulnerability to pollution: Current status and challenges. In *Earth-Science Reviews*, 185:901-927. Elsevier BV. <https://doi.org/10.1016/j.earscirev.2018.08.009>

Maqsoom, A., Aslam, B., Khalil, U., Ghorbanzadeh, O., Ashraf, H., Tufail, R.F., Farooq, D., Blaschke, T. (2020). A GIS-based DRASTIC model and an adjusted DRASTIC model (DRASTICA) for groundwater susceptibility assessment along the China–Pakistan Economic Corridor (CPEC) route. *ISPRS International Journal of Geo-Information*, 9(5), article 332. <https://doi.org/10.3390/ijgi9050332>

Malan H., Bath A., Day A. & Joubert A. (2003). A simple flow-concentration modelling method for integrating water quality and water quantity in rivers. *Water SA*, 29(3):305-312. <https://doi.org/10.4314/wsa.v29i3.4932>

Map of Welkom. (2023). Google Maps. <https://www.google.com/maps/place/Welkom/@->

[28.004114,26.6182978,11z/data=!3S6!4b1!4m6!3m5!1s0x1e910cba46a59d83:0x6f8e9cb55017f815!8m2!3d-27.9868156!4d26.7557854!16zL20vMDR4ZDd5?entry=ttu](https://doi.org/10.1007/s13202-023-01611-8)

[Accessed 31 January 2023].

Margat, J. (1968). Vulnérabilité des nappes d'eau souterraine à la pollution. Bases de al cartographie [Groundwater vulnerability to contamination]. BRGM, 68 SGL 198 HYD. Orlean, France.

Masindi, V. and Muedi, K.L. (2018). Environmental contamination by heavy metals. In: Saleh, H.E-D.M. and Aglan, R.F. (Eds.), *Heavy metals* (pp. 115-133). Intech Open. <https://doi.org/10.5772/intechopen.76082>

Maynard, J.B. and Klein, G.D. (1995). Tectonic subsidence analysis in the characterization of sedimentary ore deposits: Examples for the Witwatersrand (Au), White Opine (Cu), and Molango (Mn). *Economic Geology*, 90(1):37-50. <https://doi.org/10.2113/gsecongeo.90.1.37>

Mazdarani, A., Kadkhodaie, A., Wood, D.A. and Soluki, Z. (2023). Natural fractures characterization by integration of FMI logs, well logs and core data: A case study from the Sarvak Formation (Iran). *Journal of Petroleum Exploration and Production Technology*, 13(5):1247-1263. <https://doi.org/10.1007/s13202-023-01611-8>

McCarthy, T. (2006). The Witwatersrand Supergroup. In: Johnson, M.R., Anhaeusser, C.R. and Thomas, R.J. (Eds.), *The Geology of South Africa* (pp. 155-186). Johannesburg: Council for Geoscience.

McCarthy, T.S. (2011). The impact of acid mine drainage in South Africa. *South African Journal of Science*, 107(5/6). <https://doi.org/10.4102/sajs.v107i5/6.712>

McCarthy, T.S., Corner, B., Lombard, H., Beukes, N.J., Armstrong, R.A. and Cawthorn, R.G. (2018). The pre-Karoo geology of the southern portion of the Kaapvaal Craton, South Africa. *Journal of Geology*, 121(1):1-22. <https://doi.org/10.25131/sajg.121.0006>

Megha, P.U., Kavya, P., Murugan, S., and Harikumar, P.S. (2015). Sanitation mapping of groundwater contamination in rural village of India. *Journal of Environmental Protection*, 6(1):34-44. <http://dx.doi.org/10.4236/jep.2015.61005>

Mester, T., Szabó, G., Sajtos, Z., Baranyai, E., Szabó, G. and Balla, D. (2022). Environmental hazards of an unrecultivated liquid waste disposal site on soil and groundwater. *Water (Switzerland)*, 14(2), article 226.

<https://doi.org/10.3390/w14020226>

Machiwal, D., Jha, M.K., Singh, V.P. and Mohan, C. (2018). Assessment and mapping of groundwater vulnerability to pollution: Current status and challenges. *Earth-Science Reviews*, 185:901-927. <https://doi.org/10.1016/j.earscirev.2018.08.009>

Misi, A., Gumindoga, W. and Hoko, Z. (2018). An assessment of groundwater potential and vulnerability in the Upper Manyame sub-catchment of Zimbabwe. *Physics and Chemistry of the Earth*, 105:72-83. <https://doi.org/10.1016/j.pce.2018.03.003>

Moges, S.S. and Dinka, M.O. (2021). Assessment of groundwater vulnerability using the DRASTIC model: A case study of quaternary catchment A21C, Limpopo River Basin, South Africa. *Journal of Water and Land Development*, 49:35-46.

<https://doi.org/10.24425/jwld.2021.137094>

Moret, W. (2014). *Vulnerability assessment methodologies: A review of the literature*. https://www.marketlinks.org/sites/default/files/resource/files/VulnerabilityAssessmentMethodologies_LiteratureReview_2ndEd.pdf

Mosuro, G.O., Omosanya, K.O., Bayewu, O.O., Oloruntola, M.O., Laniyan, T.A., Atobi, O., Okubena, M., Popoola, E. and Adekoya, F. (2017). Assessment of groundwater vulnerability to leachate infiltration using electrical resistivity method. *Applied Water Science*, 7(5):2195-2207. <https://doi.org/10.1007/s13201-016-0393-4>

Musekiwa, C. and Majola, K. (2011). *Groundwater vulnerability map of South Africa*. Council for Geoscience Report number: 2011-0063. Bellville, South Africa

Musekiwa, C. and Majola, K. (2013). Groundwater vulnerability map for South Africa. *South African Journal of Geomatics*, 2(2):152-163.

<https://www.sajg.org.za/index.php/sajg/article/view/79/46>

Naicker, K., Cukrowska, E. and McCarthy, T.S. (2003). Acid mine drainage arising from gold mining activity in Johannesburg, South Africa and environs. *Environmental Pollution*, 122(1):29-40. [https://doi.org/10.1016/S0269-7491\(02\)00281-6](https://doi.org/10.1016/S0269-7491(02)00281-6)

Nemaxwi, P., Odiyo, J.O. and Makungo, R. (2019). Estimation of groundwater recharge response from rainfall events in a semi-arid fractured aquifer: Case study of quaternary catchment A91H, Limpopo Province, South Africa. *Cogent Engineering*, 6(1), p.1635815. <https://doi.org/10.1080/23311916.2019.1635815>

Neukum, C and Azzam, R. (2009). Quantitative assessment of intrinsic groundwater vulnerability to contamination using numerical simulations. *Science of the Total Environment*, 408(2):245-254. <https://doi.org/10.1016/j.scitotenv.2009.09.046>

Neukum, C., Hötzl, H., Himmelsbach, T. (2008). Validation of vulnerability mapping methods by field investigations and numerical modelling. *Hydrogeology Journal*, 16, 641–658. <https://doi.org/10.1007/s10040-007-0249-y>

Ngure, V., Davies, T., Kinuthia, G., Sitati, N., Shisia, S. and Oyoo-Okoth, E. (2014). Concentration levels of potentially harmful elements from gold mining in Lake Victoria Region, Kenya: Environmental and health implications. *Journal of Geochemical Exploration*, 144(Part C):511-516. <https://doi.org/10.1016/j.gexplo.2014.04.004>

NGWA (National Ground Water Association). (1999). *Unconfined or water table aquifers*. <https://www.ngwa.org/what-is-groundwater/About-groundwater/unconfined-or-water-table-aquifers>

Nichols, G. (2009). *Sedimentology and Stratigraphy*. Wiley-Blackwell.

Niedbalska, K. (2022). Parametric assessment of groundwater vulnerability to pollution within an open pit reclaimed by gangue. In *Applied Water Science*, 12(12), article 261. <https://doi.org/10.1007/s13201-022-01783-4>

Nimer, H.S. (2020). Effects of extreme rainfall events on sediment and contaminants. Doctoral thesis. Department of Civil and Environmental Engineering, Northeastern University Boston, Massachusetts.

<https://repository.library.northeastern.edu/files/neu:bz60cx08k/fulltext.pdf>

Nkuba, B., Bervoets, L. and Geenen, S. (2019). Invisible and ignored? Local perspectives on mercury in Congolese gold mining. *Journal of Cleaner Production*, 221:795-804. <https://doi.org/10.1016/j.jclepro.2019.01.174>

Ochieng, G.M., Seanego, E.S. and Nkwonta, O.I. (2010). Impacts of mining on water resources in South Africa: A review. *Scientific Research and Essays*, 5(22):3351-3357. https://academicjournals.org/article/article1380621809_Ochieng%20et%20al.pdf

Odonkor, S.T. and Ado, K.K. (2013). Bacteriological profile and physico-chemical quality of ground water: A case study of bore hole water sources in a rural Ghanaian community. *International Journal of Current Microbiology and Applied Sciences*, 2(8):21-40. <https://www.ijcmas.com/vol-2-8/Stephen%20T.%20Odonkor%20and%20Kennedy%20K.%20Addo.pdf>

Oke, S.A. (2015). Evaluation of the vulnerability of selected aquifer systems in the Eastern Dahomey Basin, South Western Nigeria. Doctoral thesis, University of the Free State, Bloemfontein. <http://hdl.handle.net/11660/2357>

Oke, S.A. (2020). Regional aquifer vulnerability and pollution sensitivity analysis of drastic application to Dahomey basin of Nigeria. *International Journal of Environmental Research and Public Health*, 17(7), article 2609. <https://doi.org/10.3390/ijerph17072609>

Oke, S.A., Vermeulen, D. and Gomo, M. (2016). Aquifer vulnerability assessment of the Dahomey Basin using the RTt method. *Environmental Earth Sciences*, 75, article 964. <https://doi.org/10.1007/s12665-016-5792-1>

Oke, S.A., Vermeulen, D. and Gomo, M. (2018). Intrinsic vulnerability assessment of shallow aquifers of the sedimentary basin of southwestern Nigeria. *Jamba: Journal of Disaster Risk Studies*, 10(1), article 333. <https://doi.org/10.4102/jamba.v10i1.333>

Okeke, O.C. and Okogbue, C.O. (2011). Shales: A review of their classifications, properties and importance to the petroleum industry. *Global Journal of Geological Sciences*, 9(1): 55-73. <https://www.ajol.info/index.php/gjgs/article/view/79252/69559>

Okereafor, G., Makhatha, M., Mekuto, L. and Mavumengwana, V. (2019). Evaluation of trace elemental levels as pollution indicators in an abandoned gold mine dump in

Ekurhuleni area, South Africa. In: Murillo-Tovar, M.A., Saldarriaga-Noreña, H. and Saeid, A. (Eds.), *Trace metals in the environment-new approaches and recent advances*. IntechOpen. <https://doi.org/10.5772/intechopen.89582>

Olobatoke, R.Y. and Mathuthu, M. (2016). Heavy metal concentration in soil in the tailing dam vicinity of an old gold mine in Johannesburg, South Africa. *Canadian Journal of Soil Science*, 96(3):299-304. <https://doi.org/10.1139/cjss-2015-0081>

Oregon Department of Environmental Quality Laboratory and Environmental Assessment. (1970). *Oregon Water quality index*. <http://www.oregon.gov/deq/wq/Pages/WQI.aspx>

Oyiboka, I.J. (2014). Effects of landfill sites on groundwater quality in Igando, Alimosho Local Government Area, Lagos state. Master's dissertation, University of South Africa. <http://hdl.handle.net/10500/18733>

Palladino, M., Gennari, F., Campanaro, A., and Bosco, S. (2016). Modelling pesticide transport in agricultural landscapes using the source-pathway-receptor model. *Science of the Total Environment*, 543, 236-244.

Patel, P., Mehta, D. and Sharma, N. (2022). A review on the application of the DRASTIC method in the assessment of groundwater vulnerability. *Water Supply*, 22(5):5190-5205). <https://doi.org/10.2166/ws.2022.126>

Paun, I., Cruceru, L.V. and Chiriac, F.L. (2016). Water quality indices – Methods for evaluating the quality of drinking water. INCD ECOIND — SIMI 2016, *The environment and the industry*, Proceedings of the International Symposium, pp. 395-402. <https://doi.org/10.21698/simi.2016.0055>

Peel, M., Finlayson B. and McMahon, T. (2007). Updated world map of the Köppen-Geiger climate classification. *Hydrology and Earth System Sciences*, 11(5):1633–1644. <https://doi.org/10.5194/hess-11-1633-2007>

Population Hub. (2023). Population of Welkom. <https://population-hub.com/en/za/population-of-welkom-5756.html> [Accessed, 30 January 2023]

Pouye, A., Faye, S.C., Diédhiou, M., Gaye, C.B. and Taylor, R.G. (2022). An evaluation of groundwater vulnerability assessment methods in a rapidly urbanizing city: Evidence from Dakar, Senegal. *Environmental Earth Sciences*, 81, article 410.

<https://doi.org/10.1007/s12665-022-10531-5>

Pretorius, D. (1974). The nature of the Witwatersrand gold-uranium deposits. *Economic Geology Research Unit Information Circular*, 86, p. 50.

Putranto, T.T. and Yusrizal, M.B.S. (2018). Determining the groundwater vulnerability using the aquifer vulnerability index (AVI) in the Salatiga groundwater basin in Indonesia. *AIP Conference Proceedings*, 2021, 030016 (2018). <https://doi.org/10.1063/1.5062740>

Putranto, T.T., Santi, N., Widiarso, D.A. and Pamungkas, D. (2018). Application of aquifer vulnerability index (AVI) method to assess groundwater vulnerability to contamination in Semarang urban area. *MATEC Web of Conferences*, 159.

<https://doi.org/10.1051/mateconf/201815901036>

Rahman, A. (2008) A GIS based DRASTIC model for assessing groundwater vulnerability in shallow aquifer in Aligarh, India. *Applied Geography*, 28(1):32-53.

<https://doi.org/10.1016/j.apgeog.2007.07.008>

Ratikane, M. (2013). Quality of drinking water sources in the Bloemfontein area of the Mangaung Metropolitan Municipality. Master's dissertation, Central University of Technology, Bloemfontein. <http://hdl.handle.net/11462/210>

Reading, H. G. (1996). *Sedimentary Environments: Processes, Facies, and Stratigraphy*. Blackwell Publishing.

Ribeiro, L., Pindo, J.C. and Dominguez-Granda, L. (2017). Assessment of groundwater vulnerability in the Daule aquifer, Ecuador, using the susceptibility index method. *Science of the Total Environment*, 574:1674-1683.

<https://doi.org/10.1016/j.scitotenv.2016.09.004>

Richts, A. and Vrba, J. (2016). Groundwater resources and hydroclimatic extremes: Mapping global groundwater vulnerability to floods and droughts. *Environmental Earth Sciences*, 75, article 926. <https://doi.org/10.1007/s12665-016-5632-3>

Rickwood, C. and Carr, G.M. (2007). Global drinking water quality index development and sensitivity analysis report. Burlington, Ontario: United Nations Environment Programme Global Environment Monitoring System (GEMS) Water Programme.

<https://wedocs.unep.org/bitstream/handle/20.500.11822/12214/gwqi.pdf>

Ritchie, J.C. and Schiebe, F.R. (2000). Water quality. In: Schultz, G.A. and Engman, E.T. (Eds.), *Remote sensing in hydrology and water management*. Berlin, Heidelberg: Springer. https://doi.org/10.1007/978-3-642-59583-7_13

Rodriguez-Galiano, V.F., Mendes, P.M., Garcia-Soldado, M.J. Chica-Olmo, M. and Ribeiro, L. (2014). Predictive modeling of groundwater nitrate pollution using random forest and multisource variables related to intrinsic and specific vulnerability: A case study in an agricultural setting (Southern Spain). *Science of the Total Environment*, 476-477:189-206. <https://doi.org/10.1016/j.scitotenv.2014.01.001>

Ruzvidzo, S. (2019). Effects of evapotranspiration on water quality and quantity along the Modder River catchment. Central University of Technology. Available at: <http://ir.cut.ac.za/bitstream/handle/11462/2380/Ruzvidzo%2C%20Silent.pdf>

Sahu, I., Prasad, A.D. and Ahmad, I. (2022). Groundwater vulnerability assessment using SINTACS model and GIS: A case study in Raipur city. *IOP Conference Series: Earth and Environmental Science*, 982, 012070. <https://doi.org/10.1088/1755-1315/982/1/012070>

Saidi, S., Bouri, S., Dhia, H.B. and Anselme, B. (2011). Assessment of groundwater risk using intrinsic vulnerability and hazard mapping: Application to Souassi aquifer, Tunisian Sahel. *Agricultural Water Management*, 98(10):1671-1682.

<https://doi.org/10.1016/j.agwat.2011.06.005>

Sakala, E., Fourie, F., Gomo, M., Coetzee, H. and Magadaza, L. (2016). Specific groundwater vulnerability mapping: Case study of acid mine drainage in the Witbank Coalfield, South Africa. Proceedings of the 6th IASTED International Conference on Environment and Water Resource Management, Africa EWRM 2016, 93-100.

<https://doi.org/10.2316/P.2016.836-010>

Salako, A.O and Adepelumi, A.A. (2018). Aquifer, classification and characterization. In: Javaid, M.S. and Khan, S.A. (Eds.), *Aquifers – Matrix and Fluids*. InTech. <https://doi.org/10.5772/intechopen.72692>

SANS (South African National Standards). SANS 241. *Guidelines for drinking water quality*. 2nd edition (2015). South African Bureau of Standards.

https://gwd.org.za/sites/default/files/2022-05/SANS241ED7_TC147_DSS_1%20-%20draft%20May%202022.pdf

Sarkar, C. and Abbasi, S.A. (2006). Qualidex – A new software for generating water quality indices. *Environmental Monitoring and Assessment*, 119(1-3):201-231/

<https://doi.org/10.1007/s10661-005-9023-6>

Satheeskumar, V., Subramani, T., Lakshumanan, C., Roy, P. D. and Karunanidhi, D. (2020). Groundwater chemistry and demarcation of seawater intrusion zones in the Thamirabarani delta of south India based on geochemical signatures. *Environmental Geochemistry and Health*, 43:757-770. <https://doi.org/10.1007/s10653-020-00536-z>

SAWS (South African Weather Service). (2023). *Annual state of the climate of South Africa 2022*. Pretoria, South Africa.

https://www.weathersa.co.za/Documents/Corporate/Annual%20State%20of%20the%20Climate%202023_02042024155932.pdf

Schlumberger. (2023). Petrophysics and Formation Evaluation. Schlumberger Oilfield Glossary.

Seo, M., Lee, H. and Kim, Y. (2019). Relationship between Coliform bacteria and water quality factors at Weir stations in the Nakdong river, South Korea. *Water*, 11(6), article 1171; <https://doi.org/10.3390/w11061171>

Shah, S.H.I.A., Yan, J., Ullah, I., Aslam, B., Tariq, A., Zhang, L. and Mumtaz, F. (2021). Classification of aquifer vulnerability by using the drastic index and geo-electrical techniques. *Water (Switzerland)*, 13(16), article 2144.

<https://doi.org/10.3390/w13162144>

South African Committee for Stratigraphy. (1980). *Stratigraphy of South Africa, Part 1: "Randian Erathem"*. Pretoria: Geological Survey South Africa, Pretoria.

<https://www.stratigraphy.org.za/>

South African Venues. (2011). www.SA.venues.com (accessed, July 4, 2023).

Still, D. and Nash, S. (2005). Groundwater contamination due to pit latrines located in a sandy aquifer: A case study from Maputaland. Report to the Water Research Commission by Partners in Development. (WRC Report No. TT 527/12).

Strategic Assessment of Florida's Environment. 1995. *Florida stream water quality index, statewide summary*. Available at:

<http://www.pepps.fsu.edu/safe/environ/swq1.html>

Sukmawati Rukmana, B.T., Bargawa, W.S. and Cahyadi, T.A. (2020). Assessment of Groundwater Vulnerability Using GOD Method. *IOP Conference Series: Earth and Environmental Science*, 477, 012020. <https://doi.org/10.1088/1755-1315/477/1/012020>

Taghavi, N., Niven, R.K., Kramer, M. and Paull, D.J. (2023). Comparison of DRASTIC and DRASTICL groundwater vulnerability assessments of the Burdekin Basin, Queensland, Australia. *Science of the Total Environment*, 858, 159945.

<https://doi.org/10.1016/j.scitotenv.2022.159945>

Thapinta, A. and Hudak, P.F. (2003). Use of geographic information systems for assessing groundwater pollution potential by pesticides in Central Thailand. *Environment International*, 29(1):87-93. [https://doi.org/10.1016/s0160-4120\(02\)00149-6](https://doi.org/10.1016/s0160-4120(02)00149-6)

Tucker, M. E. (1991). *Sedimentary Petrology: An Introduction to the Origin of Sedimentary Rocks*. Blackwell Science.

Tucker, M.E. (2001). *Sedimentary Petrology: An Introduction to the Origin of Sedimentary Rocks*. Blackwell Science.

Tutu, H., McCarthy, T. S. and Cukrowska, E. (2008). The chemical characteristics of acid mine drainage with particular reference to sources, distribution and remediation: The Witwatersrand Basin, South Africa as a case study. *Applied Geochemistry*, 23(12), 3666-3684. <https://doi.org/10.1016/j.apgeochem.2008.09.002>

Van Stempvoort, D., Ewert, L., and Wassenaar, L. (1992). AVI: A method for Groundwater Protection mapping in the Praire Province of Canada. PPWB Report No. 114. National Hydrology Research Institute, Saskatoon Saskatchewan, Canada.

Verhille, S. (2013). Understanding microbial indicators for drinking water assessment: Interpretation of test results and public health significance. National Collaborating Centre for Environmental Health.

https://www.ncceh.ca/sites/default/files/Microbial_Indicators_Jan_2013_0.pdf

Verlicchi, P. and Grillini, V. (2020). Surface water and groundwater quality in South Africa and Mozambique – Analysis of the most critical pollutants for drinking purposes and challenges in water treatment selection. *Water* 2020, 12(1), article 305.

<https://doi.org/10.3390/w12010305>

Vrba, J. and Zaporozec, A. (1994). Guidebook on mapping groundwater vulnerability. A A Balkema Publishers.

Vu, T.D., Ni, C.F., Li, W.C. and Truong, M.H. (2019). Modified index-overlay method to assess spatial-temporal variations of groundwater vulnerability and groundwater contamination risk in areas with variable activities of agriculture developments. *Water (Switzerland)*, 11(12), article 2492. <https://doi.org/10.3390/w11122492>

Wade, P.W., Woodbourne, S., Morris, W.M., Vos, P. and Jarvis, N.V. (2002). Tier 1 risk assessment of radionuclides in selected sediments of the Mooi River. Water Research Commission (WRC) Report, No. K5/1095, Pretoria. <https://www.wrc.org.za/wp-content/uploads/mdocs/1095-1-021.pdf>

Wang, H., Gao, G.E., Zhang, M-J, Li, X-H, Zhang, S-L. and Jia, L-Z. (2015). Effects of rainfall intensity on groundwater recharge based on simulated rainfall experiments and a groundwater flow model. *CATENA*, 127:80-91.

<https://doi.org/10.1016/j.catena.2014.12.014>

Wang, X., Jia, S., Liu, Z. and Mao, B. (2022). Watershed-scale shallow groundwater anthropogenic nitrate source, loading, and contamination assessment in a typical wheat production region: Case Study in Yiluo River watershed, middle of China. *Water (Switzerland)*, 14(23), article 3979. <https://doi.org/10.3390/w14233979>

Water Research Commission. (2017). Groundwater sampling manual. (WRC Report No. TT 733/17). Pretoria, South Africa. <https://www.wrc.org.za/wp-content/uploads/mdocs/TT%20733-17.pdf>

Wei, A., Bi, P., Guo, J., Lu, S. and Li, D. (2021). Modified DRASTIC model for groundwater vulnerability to nitrate contamination in the Dagujia river basin, China. *Water Science and Technology: Water Supply*, 21(4):1793-1805. <https://doi.org/10.2166/WS.2021.018>

Wertz, T.A. & Shank, M.K. (2019). Land use from water quality: Development of a water quality index across Pennsylvania streams. *Ecosphere*, 10(11), article e02947. <https://doi.org/10.1002/ecs2.2947>

Whitehead, P., Wade, A.J. and Butterfield, D. (2009). Potential impacts of climate change on water quality and ecology in six UK rivers. *Hydrology Research*, 40(2-3):113-122. <https://doi.org/10.2166/nh.2009.078>

WHO (World Health Organization). 2018. A global overview of national regulations and standards for drinking-water quality. Geneva. <https://apps.who.int/iris/bitstream/handle/10665/272345/9789241513760-eng.pdf>

WHO (World Health Organization). (2022). *Guidelines for drinking-water quality: Fourth edition incorporating the first and second addenda*. Geneva. <https://iris.who.int/bitstream/handle/10665/352532/9789240045064-eng.pdf>

Wong, J.W.C., Lai, K.M., Su, D.S. and Fang, M. (2001). Availability of heavy metals for *Brassica Chinensis* grown in an acidic loamy soil amended with domestic and industrial sewage sludge. *Journal of Water, Air and Soil Pollution*, 128:339-353. <https://doi.org/10.1023/A:1010353107240>

Wu, J., Teng, Y., Lu, S., Wang, Y. and Jiao, X. (2014). Evaluation of soil contamination indices in a mining area of Jiangxi, China. *PLoS ONE*, 9(11), article e112917. <https://doi.org/10.1371/journal.pone.0112917>

Zhang, Q., Wu, Y., & Liu, S. (2017). Agricultural pesticide contamination in watersheds: An application of the source-pathway-receptor model. *Environmental Pollution*, 229, 587-595.

Zhou, T., Li, Z., Zhang, F., Jiang, X., Shi, W., Wu, L., Christie, P. (2016). Concentrations of arsenic, cadmium and lead in human hair and typical foods in eleven Chinese cities. *Environmental Toxicology Pharmacology*, 48:150-156.

<https://doi.org/10.1016/j.etap.2016.10.010>

Zhou, Y., Li, P., Chen, M., Dong, Z. and Lu, C. (2020). Groundwater quality for potable and irrigation uses and associated health risk in southern part of Gu'an County. North China Plain. *Environmental Geochemistry and Health*, 43:813-835.

<https://doi.org/10.1007/s10653-020-00553-y>

Zwahlen, F. (Ed.) (2004). *COST action 620 – Vulnerability and risk mapping for the protection of carbonate (karst) aquifers – Final report*. European Cooperation in Science and Technology, Brussels.

<http://bookshop.europa.eu/uri?target=EUB:NOTICE:KINA20912:EN:HTML>