The Impact of the Leachate from the Botshabelo Non-Engineered Landfill on Groundwater, Surface Water and Soil Quality

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Declaration

I, Moeketsi Sesing, declare that this dissertation submitted for the Master of Health and Environmental Sciences degree in the Department of Life Sciences, Faculty of Health and Environmental Sciences at the Central University of Technology, Bloemfontein, South Africa, is my own work that has not been submitted to any other institution of higher education and training. I further declare that all the data sources used in this dissertation have been outlined in the in-text references and also in the reference list provided.

Signature of the student

Date

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In the Mighty Name of the King, the Lord Jesus Christ, the Creator of the heavens and the earth, to whom is the Kingdom, the Glory, and the Power.

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Abstract

Introduction: Many countries around the world, including South Africa, faced significant challenges in managing urban solid waste. As such, the dumping of waste has been a constant phenomenon in the landfill sites that has impacted the environment. This has resulted in a decline in groundwater, surface water, and soil quality resources that are necessary for South Africa's overall development.

Aim: This study sought to explore the effects of leachate from the Botshabelo non-engineered landfill on groundwater, surface water, and soil quality.

Methodology: Four groundwater, two soil and surface water, and one leachate samples were gathered for sampling in wet and dry seasons. The samples were examined for physicochemical characteristics and contrasted with domestic and global norms. The use Inductively Coupled Plasma Mass Spectrometry (ICP-MS) and in-situ pH analyses were conducted for water samples, Wavelength Diffraction X-Ray Flourescence (WDXRF) and X-Ray Diffraction (XRD) for soil samples. Microsoft Excel was used for statistics, and Grapher Software for geochemical evolution of the water. The complicated data were simplified using both water quality indices and soil pollution indices which gave numerical representation of the qualities with subsequent classifications as well as parameters that posed contamination. Groundwater vulnerability was conducted to outline boreholes that were most susceptible to pollution around the landfill site and was plotted using the Geographical Information System (GIS) software.

Results and discussion: The leachate was highly contaminated with Na, Cl, alkalinity, and total hardness. The mean EC for groundwater surpassed the DWAF standard and it was classified as hard. Only Mg surpassed the WHO and SANS limits. Surface water showed little to no pollution in all seasons and was moderately hard. The geochemistry of water samples revealed $CaMgHCO_3$ and CaMgCl, dominant cation, and magnesium types of water. The water−rock interaction and carbonate weathering influenced the water chemistry in the landfill site. Acidic soil polluted with vanadium and chromium was discovered.

Acording to pollution indices Mg, manganese, and uranium as contaminant sources in groundwater in all seasons. Surface water had only manganese. Groundwater was between poor and unsuitable except for surface water. Furthermore, contamination by vanadium, arsenic, copper, and chromium was found in soil, deeming it as severely contaminated. Groundwater vulnerability methods depicted very low and moderate vulnerabilities across the boreholes.

Conclusion: The study found that the Botshabelo non-engineered landfill site had minor impact on water quality due to slightly higher contamination of certain parameters, while the soil was heavily impacted. Therefore, frequent monitoring of water and soil quality at the landfill site, with the installation of leachate collection systems, bottom liners, borehole drilling, and covering of opened boreholes are recommended. These will ensure monitoring, protection, and generation of more data.

Keywords: landfilling, physicochemical parameters, water quality index, soil quality index, geochemistry, groundwater vulnerability

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List of Abbreviations

ANOVA	Analysis of Variance
AVI	Aquifer Vulnerability Index
BH	Borehole
BOD	Biological oxygen demand
CCME	Canadian Council of Ministers of the Environment
CF	Contamination Factor
COD	Chemical Oxygen Demand
DEA	Department of Environmental Affairs
DO	Dissolved Oxygen
DWAF	Department Of Water Affairs and Forestry
DWQI	Drinking Water Quality Index
DWS	Department of Water and Sanitation
EC	Electrical Conductivity
E. coli	Escherichia coli
GB	Geochemical Background
GIS	Geographic Information System
GOD	Groundwater Occurrence or Aquifer Type, Overall Lithology of Aquifer, and Depth of Groundwater
ICP	Inductively Coupled Plasma
ICP-OES	Inductively Coupled Plasma Optical Emission Spectroscopy
ICP−MS	Inductively Coupled Plasma Mass Spectrometry
IGS	Institute for Groundwater Studies
IWMP	Integrated Waste Management Plan
MOSS	Mangaung Metropolitan Open Space System
NCI	Nemerow Comprehensive Index
NPI	Nemerow Pollution Index

NSF	National Sanitation Foundation
NTU	Nephelometric Turbidity Unit
PSD	Particle Size Distribution
RTt	Rainfall Travel Time
SANS	South African National Standards
SAR	Sodium Adsorption Ratio
SRDD	Scottish Research Development Department
SW	Surface Water
TDS	Total Dissolved Solids
TOC	Total Organic Carbon
USEPA	United States Environmental Protection Agency
WAWQI	Weighted Arithmetic Water Quality Index
WHO	World Health Organization
WQI	Water Quality Index

List of Chemical Symbols

Al	Aluminium
As	Arsenic
Ba	Barium
Br	Bromide
Ca	Calcium
Cd	Cadmium
Cl^-	Chloride
Co	Cobalt
Cr	Chromium
Cu	Copper
Fe	Iron
Hg	Mercury
K	Potassium
Mn	Manganese
Mg	Magnesium
Mo	Molybdenum
Na	Sodium
NO_2	Nitrogen dioxide
NO_3	Nitrogen trioxide
Ni	Nickel
Pb	Lead
Sb	Antimony
Se	Selenium
Si	Silicon
SO_4	Sulphate
U	Uranium
V	Vanadium
Zn	Zinc

List of Measuring Units

°C	Degrees Celsius
%	Percentage
cm^3	Cubic centimetre
kg	Kilogram
km	Kilometre
km^2	Square kilometre
ℓ/s	Litre per second
m	Metre
mℓ	Millilitre
mg/ℓ	Milligram per litre
mS/m	Millisiemens per metre
mm	Millimetre
µm	Micrometre

Chapter 1
Introduction and Background

1.1 Introduction

This chapter provides a summary of South Africa's water resources, as well as landfilling in South Africa and its influence on the environment through the release of leachate into the ecosystem. The landfill location being focused on is the Botshabelo municipal solid waste landfill that is operated in the Mangaung Metropolitan Municipality in the Free State province. The chapter outlines the study questions that it seeks to answer and concludes by delineating the objectives that are addressed.

1.2 Background

1.2.1 Overview of South Africa's water resources and management

Water is the most crucial resource for sustaining life. Water has always played and continues to play an important part in the economic growth and development of many nations, and early civilisations were interested in the improvement and management of water resources to meet the immediate need. Globally, countries that have invested in water infrastructure have seen their economies secured and protected from climate change that causes havoc, such as floods and droughts (Manase, 2009). South Africa has also been successful in this matter because its economy has been able to stand drastic weather shocks due to its water storage plan, which has been in a steady incline (Manase, 2009).

Though water occurs everywhere it has always been defined by its quantity, quality and availability (Molobela and Sinha, 2011). South Africa is a water-scarce country, with an average annual rainfall of 450 mm (millimetres), less than the global average of 860 mm (Basson, 2011). Furthermore, South Africa is characterised by high rates of evaporation due to its semi-arid climate and the lack of significant rivers, and the overall runoff has a total of 49 billion cubic metres per year, which is smaller than the Zambezi River next to it. According to Kahinda and Boroto (2009), the reduction of surface water is greater than the average rainfall ranging at about 1 500−2 500 mm, due to evaporation. The natural water and groundwater availability are poor and water flow is not distributed evenly. To put it into perspective, 60% of water flow or runoff comes only from the 20% land surface as a result of uneven topography and rainfall, and the four main rivers of the country share its water with neighbouring countries.

South Africa depends mainly on surface water for industrial, urban and irrigation needs. According to Kahinda and Boroto (2009), the country allocates 60% of its water to irrigation, 23% to the urban sector, and 15% to the other sectors combined. With increasing development, coupled with population growth, there is an increase in pressure on water resources that triggers water scarcity problems. It has been estimated that by the year 2025, many South Africans will live in water scarcity conditions as current water usage is extremely high and the degree of usage is greater than the recharge. The National Treasury (2011) asserted that the urban areas (metros and large cities) consume enormous amounts of water; therefore, there is a need to consider rural areas to ensure food security. As much as there is an extensive use of surface water, its quality has been deteriorating as a result of effluents that have been deposited into them. It has been discovered that the major pollutants of surface water include agricultural runoff, industries, mining, sanitation services, and rural settlements (Kahinda and Boroto, 2009). This makes surface water unfit for use in its many applications.

On a global level, groundwater is said to be the earth's source of stored freshwater, containing about 94% of the global freshwater resource for socio-economic and environmental developments (Knüppe, 2011). In South Africa, groundwater is used for drinking, sanitation, agricultural, irrigation, and industrial sectors, and poverty reduction as a way of strategic support owing to declining surface water resources. According to the Department of Water Affairs and Forestry [DWAF] (DWAF, 2002), groundwater contributes about 15% of the nation's total volume of water and more than 280 cities and towns depend solely on groundwater, and this has increased sharply. Similar to surface water, the groundwater budget consists of a 64% allocation for irrigation, while 8% is used for domestic and mining purposes (Knüppe, 2011). The groundwater system in South Africa is largely influenced by geology and climatic conditions that directly control the rate of its extraction. The country's geohydrology is characterised by aquifers that are fractured and have zero porosity (Knüppe, 2011). This then limits the quantity that can be extracted, as only 20% of groundwater is found in major aquifers suitable for use.

The pollution/contamination of groundwater in South Africa is attributed to human activities such as land-use, mining, industrial, and solid waste disposal, which is the focus of this research. This can have a dire impact on the operation of the environment. This then calls for groundwater management and policies in order to address this worsening challenge as this threatens the quality of groundwater. However, according to the hydrogeological community, groundwater is said to be of less value and use, and therefore, this has led to mismanagement and implementation of policies and legislation (Knüppe, 2011).

However, South Africa's water resource management has improved. For example, water legislation, improved infrastructure, administration, and technologies have all contributed to the advancement of the water management systems (Basson, 2011). The National Water Act [NWA] 36 of 1998 established the concepts of fairness, sustainability, and efficiency as the foundation for water resource policy and management. The objective of the NWA is to accomplish sustainable use, protection, development, conservation, management, and control of water resources. It should be emphasised that the NWA falls under the Department of Water and Sanitation (DWS), which has overarching control for both groundwater and surface water resources. Although there are guidelines, regulations, and legislation that appear good on paper, it stands to reason that there is poor application and enforcement of such policies in all levels of management.

Furthermore, the mishandling of water resources has been a major issue in many countries, halting the general development and expansion. This challenge is caused by rapid growth population, climate change, politics, stagnant economic development, financial incapacity, corruption, lack of implementation of legal frameworks, and institutional inadequacy. These challenges are also abundant in South Africa. Moreover, it is estimated that these water management problems will still be a main concern in the future as they interrupt the communication, transport, mining, agriculture, education, and health sectors (Molobela and Sinha, 2011).

1.2.2 Landfills and their environmental impact

The management of waste is a major environmental concern in South Africa, and municipalities are responsible for garbage removal under Section 156(1)(a) of the Constitution of the Republic of South Africa (1996). However, municipalities have been unable to cope with the appropriate management of solid waste due to causes such as urbanisation, population expansion, and income increases, which together have contributed to an increase in garbage generation (Polasi et al., 2020). This resulted in immense pressure on municipalities to deliver services concerning waste management facilities like landfills. This challenge is worsened by the municipalities' shortcomings, which include lack of planning, financial incapacity, lack of skilled labour force, poor infrastructure, and policy enforcement (Department of Environmental Affairs [DEA], 2018).

The National Waste Management Strategy (Department of Environment, Forestry, and Fisheries [DEFF], 2020) described waste hierarchy practices in terms of waste avoidance, reduction, re-use, recycling, recovery, and disposal. This practice, when applied effectively, will ensure that little to no waste is disposed of in the landfills; hence, this would reduce the quantity of landfills around the country by a large sum. When landfilling is reduced it means that its impacts on the environment are reduced and the municipalities will have least concerns about waste management as a whole. It is regrettable to say that municipalities have not been able to fulfil this mandate, and this has resulted in an increased practice of landfilling in the country.

Landfill is defined as a physical place where municipal solid waste is dumped in a controlled manner (Osazee, 2021). This method of waste disposal has been actively practiced in the history of mankind and it is common globally as the generally accepted approach for waste management. Both industrialised and developing countries are currently using this cost-effective, user-friendly, and less difficult method (Mishra et al., 2020). Landfilling is poorly operated in South Africa due to challenges faced by municipalities and stakeholders; hence, DWAF (1998) has detailed eight requirement guidelines as a means to alleviate this overarching problem.

DWAF (1998) serves a responsibility of waste by the effective security of the ecosystem from the effects of undesirable waste disposal methods. It does this by taking giant steps by first inducing control systems by issuing permits for the operation of landfill sites. Also, it sets the standards needed for the use of the basic requirements for operating such dump sites. One of the requirements of the permit acquisition is a proper site to operate landfills but the current situation in the country shows that landfill areas are depleted, leading to non-availability of space to establish landfills (Chvatal and Smith, 2015). This results in illegal dumping of waste such as domestic, business, demolition, and garden wastes on the environment, which lead to contamination of groundwater and soil (Haywood et al., 2021).

Although landfilling is the most frequent way to dispose of waste, it has been shown to harm the ecosystem. According to Negi et al. (2018), the landfill design has an influence on the environment because of the leakage of harmful substances. In many developing nations worldwide landfills are devoid of a sanitary landfill classification category as they do not meet the requirements (Negi et al., 2018). Many landfills do not have the proper liners, leachate collection systems, and landfill covers, thereby posing a great risk of pollution to the environment (Idowu et al., 2019). The leachate formation in landfills is the greatest source of contamination of the environment (Abiriga et al., 2020). Leachates are liquids that contain organic and inorganic salts, trace elements, and heavy metals that have different concentrations. They are formed inside the landfill due to the combination of surplus rainwater with waste, whereby through interaction, chemical and biological processes occur and leach toxins from the waste into the resultant leachate (Negi et al., 2018). As a result, the leachate infiltrates into the subsurface to pollute groundwater and makes it unfit for consumption and use, and as soon as it is polluted, groundwater becomes extremely difficult to purify due to the difficult access coupled with long pollutant residence time and storage. Moreover, leachate can reach surface water and reduce the amount of oxygen through runoff from the landfill site. Soils are also victims of leachate contamination that emanates from the landfills.

In order to fulfil sustainable development, it is vital that the environment is protected from any anthropogenic source of pollution as much as possible because it is the only factor over which we have control, in contrast to natural pollution factors such as change. The management of both water resources (groundwater and surface water) and soil will assure water and food security for current and future generations.

1.3 Problem statement

The major repercussions of landfilling on the environment have been intensively examined around the world, in both industrialised and developing countries. Being a widely used method of waste disposal that is cost-effective, combined with its ease of use, it has been discovered as a great threat to the environment, particularly on surface water, groundwater, and soil. Magda et al. (2015) denoted that their main reason for pollution is the generation of landfill leachate, which has catastrophic consequences on the environment as it comes into contact with it. Makhadi et al. (2020) investigated the consequences of landfill leachates on the environment in Bloemfontein where they discovered that the influx of rainwater in the landfill (for example, rainy season) generates large quantities of leachate that flows as runoff into both soil and surface water, and seeps into the subsurface to reach the water table, leading to groundwater contamination. As mentioned previously, the infrastructural development of the landfill site determines its ability to pollute the environment – with poorly developed, unlined and absent leachate collection, the landfill can pollute surface water, groundwater, and soil (Idowu et al., 2019). This resulting impact is for the large part due to the inadequate management of solid waste by individual countries, and it is stated that the level of waste management of a nation reflects its economic development and growth.

South Africa is not abundant in fresh water resources and it is the 30th water resource-scarce nation in the world (Pietersen et al., 2011). Although the country uses approximately 70% of surface water for various economic development purposes, such as agriculture, mining, industrial, business, and domestic needs, it is appropriate to admit that this resource is depleting due to the huge pressure put on it. Consequently, to achieve sustainable development, investment, and economic growth, there must be a joint utilisation of both surface and groundwater resources. These resources are interconnected where they both recharge and discharge into each other (Pietersen et al., 2011), meaning that an increase in surface water supply can recharge the aquifers and the aquifers can discharge into surface water in periods of low rainfall. This interconnection means that if one gets contaminated, another will follow suit. Hence, the pollution of surface water and groundwater by landfill sites is widely studied and continues to be researched. Groundwater is able to meet water demands for rural and urban communities, as well as ensuring water and food security. In South Africa, groundwater is largely used in provinces such as the North West and KwaZulu-Natal, coupled with the Eastern Cape where more than 50% of rural communities depend on groundwater. Makhadi et al. (2020) reported that in the Free State, groundwater is used for irrigation in farming areas, and there are certain communities that use groundwater for domestic purposes such as Wepener, Dewetsdorp, Reddersburg, and Edenburg (Water Research Commission [WRC], 1994). Botshabelo is one of the most developing townships that is situated on the eastern side of Bloemfontein. It is also one of the fastest-growing townships of South Africa as people are migrating from neighbouring rural areas and towns (WRC, 1994). Surface water is the predominant resource that is supplied to the Botshabelo area by Vaal Central Water and the Botshabelo landfill site that is managed by the Mangaung Metropolitan Municipality. Because of the increasing demand to protect environmental resources, the purpose of this study is to investigate the impact of the Botshabelo non-engineered landfill site on the quality of groundwater, surface water, and soil.

1.4 Motivation of the study

The investigation of leachate pollution of groundwater, surface water, and soil resources around landfill sites is vital due to the complex and ongoing threat it poses to both environmental integrity and public health. Landfill leachate is a highly heterogeneous effluent that contains a lethal mixture of chemicals. Its migration into the surrounding environment is a primary conduit for ecosystem destruction. Elevated concentrations of major ions act as strong hydrochemical indicators or tracers, confirming leachate plume migration and defining its spatial extent in groundwater aquifers. Concurrently, the quantification of toxic trace elements is paramount, because these persistent, bioaccumulative contaminants can infiltrate the food web, posing significant risks to groundwater, surface water and soil-thereby affecting their quality for use for various purposes. It has been noted that the conventional treatment methods of leachate have been nullified due their lack of environmental and economic viability. Consequently, a thorough examination encompassing these parameters is not solely an environmental evaluation but an essential prerequisite for formulating resilient risk assessment frameworks, crafting efficient containment and remediation strategies, and guiding evidence-based policy and landfill management practices to preserve invaluable water and soil resources and safeguard human health for present and future generations.

1.5 Aim and objectives

Main aim: To explore the effects of leachate from the Botshabelo non-engineered landfill on groundwater, surface water, and soil quality.

In addressing the explanation of the problem and the questions that have been put out, the objectives of the study were as follows:

1. To assess groundwater, surface water, and soil quality using physicochemical parameters in comparison with the leachate chemistry to decipher whether there is contamination.
2. To determine groundwater, surface water, and soil quality using physicochemical parameters in comparison with the local and global standards. This is to determine how the landfill affected the water and soil qualities over two seasons.
3. To assess the geochemistry of groundwater, surface water, and soil over two seasons to understand their evolution.
4. To simplify the water and soil quality results by the application of groundwater, surface water, and soil quality indices.
5. To determine the susceptibility of groundwater to pollution by the application of Rainfall Travel Time (RTt) and the groundwater occurrence or aquifer type, overall lithology of aquifer, and depth of groundwater (Groundwater occurrence, Overlying lithology, Depth to aquifer level (GOD)) vulnerability methods, and to delineate vulnerable areas for future decision-making.

1.6 Research questions

The study intended to solve the following research questions:

1. What is the quality of the groundwater, surface water, and soil at the Botshabelo landfill site?
2. Does the landfill cause pollution in and around the landfill?
3. If contamination takes place, to what degree?
4. Are there any environmental protection measures included in the landfill or the landfill design?

5. Is groundwater vulnerable to contamination due to leachate produced in the landfill?

1.7 Scope of the study

The study looked at the condition of groundwater, surface water, and soil near the Botshabelo dump site as a result of leachate. Water and soil samples were collected and analysed to assess their physicochemical properties and suitability for drinking and irrigation. In addition, pollution indices were conducted to determine the pollution level of the water and soil. Groundwater vulnerability was carried out to delineate susceptible areas around the landfill site to emerging pollutants.

1.8 Limitations of the study

The study was constrained by a minimal amount of sampling data and sampling sites that prevented a more detailed and thoroughly researched presentation of results. This was influenced by the lack of access to the boreholes and a surface water stream that were located inside the premises of the abattoir next to the Botshabelo landfill site. The presence of these additional, missing sampling points could have added more data to increase the accuracy and credibility of the results that would, in turn, increase confidence in them. Also, the municipal waste dumping research in South Africa lacks attention by many researchers; hence, this leads to a limited amount of data that can be used for future researchers who could benefit from such data. Additionally, lack of monitoring of the landfill site by the Mangaung Metropolitan Municipality also limited the available data that could have been used for the outcomes of this study. This study, then, stands as a base in providing more knowledge and understanding for future studies.

1.9 Dissertation outline

The dissertation consists of five chapters which are outlined as follows:

Chapter 1- It is the introduction, which provides a general overview and describes South Africa’s water resources, as well as landfilling and its environmental impact. It also mentions the legislations and policies that governs both water and waste management.

Chapter 2- Literature review, which delves into much depth regarding South Africa's management of waste and the global practice at large. It also outlines the process of landfill pollution by leachates and their impact on the environment. It concludes by looking into literature on water and soil quality determinants, vulnerability studies and water quality indices.

Chapter 3- The chapter focuses exclusively on the background of the study area, delineating the geology, location, and waste management condition of the landfill site. The methodology is the second part of the chapter outlining the sampling procedure and analysis methods that are used in the study.

Chapter 4- The results and discussion is given in detail with the leachate, water, and soil quality parameters clearly shown and discussed in detail. Also, other water quality determinants, including water quality indices, geochemical speciation, groundwater vulnerability. Soil quality and pollution indices are clearly depicted by means of figures, maps and tables with descriptions.

Chapter 5- The chapter gives conclusions of the findings studied, plus recommendations on future studies on the study area.

Chapter 2
Literature Review

2.1 Introduction

The chapter provides an insight into waste management practices that have been adopted on both national and global scales. It accomplishes this by describing waste management strategies and procedures proposed by each country's governing structures, including regulatory frameworks that have been adopted to implement those strategies. This chapter also reviews the shortfalls and the poor solid waste management. Landfilling is the main anthropogenic activity being put into emphasis because of its impact on the environment, which include surface water, groundwater, and soil. To identify and measure pollution that is posed by landfilling, water and soil quality assessment is carried out to help decision-makers, researchers, and governments, as well as the public, to understand this challenge. The chapter also examines pollution indices and groundwater vulnerability methodologies to quantify the risk or sensitivity of water and soil to contamination caused by landfilling. The chapter concludes by identifying gaps that exist in literature in order to address them.

2.2 Management of solid waste

Solid waste management has been a serious global concern for humans throughout the history of the earth (Hettiarachchi et al., 2018). The management of solid waste is a worldwide concern that affects both economic sustainability and environmental damage (Ferronato and Torretta, 2019). Population growth, urbanisation, and economic development have all resulted in improved human health and well-being. As a result, consumption has increased, as has solid waste generation (Chen, 2018). As claimed by the World Bank in 2016, around 2,01 billion tons of residential garbage were generated worldwide, averaging 0,74 kg per person per day (see also Shershneva, 2022). This figure is anticipated to climb to 3,4 billion tons by 2050, however, there is a changing tendency in the management of such trash in both developed and developing nations.

2.2.1 Unsustainable management of solid wastes in developing countries

In developed countries, there are strict regulations that are coupled with their economic plans to manage solid waste (Di Maria et al., 2017). According to Di Maria et al. (2017), the effective waste management approach is defined in this order: prevention, reuse, recycle, recovery and disposal. This was put in the legislation from 1991 by first Directive 91/156/EEC on waste. The goal of this directive was for municipal solid waste to be recycled at least at 50%. The European countries, including Italy, strictly adhere to these regulations; for example, in 2015, the total quantity of waste produced was 519 kg per capita, which was greater than the national average (Di Maria et al., 2017). Furthermore, treatment facilities were put in place to separate the collected waste for the production of organic fertiliser from organic waste before being disposed at landfills as per the Italian legislation. Additionally, Abis et al. (2020) asserted that the 10% target of zero landfilling was achieved in 2018 in Belgium, Austria, Germany, Netherlands, Sweden, and Denmark using thermal treatments. Their study also demonstrated that landfilling is not the primary goal of the management of waste in European countries, but the recovery of waste to turn them into secondary raw materials for building material production, for example, construction.

The country's economic development reflects its solid waste management level. It is a known fact that an increase in the economic status, welfare, and technological advancement of a country consequently generates an increase in waste generation (Shershneva, 2022; Wilson et al., 2012). Developed nations are characterised by higher income levels than developing ones, which puts them ahead in management of waste. In their study, comparing 20 cities for waste management, Wilson et al. (2012) discovered that in high income countries, residents are able to afford waste collection fees charged by private companies as well as the government. It is demonstrated by the study that the budget per capita is $75 for high income countries as opposed to $1,4 in low income ones. Also, their results demonstrated a high degree of controlled disposal (100%) from countries such as Austria, the Netherlands, and the United States. This is shown by the state-of-the-art facilities at landfill sites that are properly designed and have leachate collection sites combined with final cover (Idowu et al., 2019). These facilities include fencing, gate control, and waste placement that decreases the risk of water and soil pollution. Digital waste management technologies are among the developments in developed countries (Shershneva, 2022).

Developing nations, in contrast, show a contrasting trend in dealing with solid waste management. This is characterised by poor or uncontrolled disposal of waste such as open dumps and landfills, poor governance, lack of regulation, corruption, poor financial planning, and lack of skilled professionals (Hettiarachchi et al., 2018). Developing nations are typical of low-income, economic vulnerability, as well as human resource weakness (Glawe et al., 2014). Similar to developed countries, developing nations also have a fast growing population and increasing rates for solid waste, but the management has not caught up with this phenomenon. Hettiarachchi et al. (2018) studied the solid waste management in the Caribbean and Latin American countries and asserted an increase in population by 80% in the cities under this region, with a total of 600 million people. This is linked with the municipal solid waste generation of 1,09 kg per capita per day, which is far more than Africa. South Africa has also seen a rise of more than three million people between 2013 and 2017, where the country saw the growth in waste generation from 108 million tons (DEA, 2011) to 121 million tons in 2017 (DEA, 2018).

Furthermore, developing countries portray poor adherence regulations in managing their waste, and in some cases, regulations are absent at all. Authorities such as the government are responsible for organising rules and form regulations to achieve common goals (Hettiarachchi et al., 2018). This is collectively done to show the stakeholders involved and who is responsible for which role; however, if such structures are not clearly defined, governance is to be probed. Sadly, this is the reality of most of the developing nations – poor governance and lack of regulations. Glawe et al. (2014) observed that in the cities of the least developed nations, landfilling is the most commonly practiced method to dispose of waste with disregard to the requirements of a sanitary landfill. According to Ferronato and Torretta (2019) and Seng et al. (2018), open dumping and poor leachate management cause environmental problems such as groundwater and surface water pollution. The 1999 National Waste Management Strategy in South Africa established a waste management policy hierarchy that includes prevention, reuse, recycling, recovery, and disposal as a last choice (DEA, 2018). However, due to a lack of governance and adherence, it is difficult to apply these regulations (Zhakata et al., 2016).

Solid waste management in developing countries is depicted by poor financing, lack of human resources, and corruption. Hettiarachchi et al. (2018) observed that municipal solid waste management services are poorly catered for because of irregular planning and structuring of urban areas during their expansion. Because of the low-income state in developing nations, the government provides the solid waste services, thereby putting pressure on financial provision and making the municipalities unable to sustain their services. Additionally, in South Africa, poor management of funds and corruption have also been reported by Nyika et al. (2020). They found that the management of solid waste was interrupted in terms of labour management, infrastructure, and equipment. In addition, these challenges have caused maintenance, employee payments, and capital expenditure to be almost impossible to meet (Nyika et al., 2020). Gutberlet (2018) reported that challenges of corruption and poor financing have led to cluttered waste collection schedules, ineffective sanitation, and unreliable services. Financial constraints have led to the partnerships of donors such as the United Nations Development Programme (UUDP) and the German government to assist local governments (Glawe et al., 2014).

2.2.2 Landfilling

For decades, several countries have embraced landfilling as a means of disposing of municipal solid waste (Adhikari et al., 2014). Mishra et al. (2020) designed it for waste storage and treatment, and it is the most simple and cost-effective way of garbage disposal compared to other varieties. Mishra et al. (2020) said that although they pose major environmental consequences on water, soil, and health, they feature various positives, which include: convenience of use, keeping cities and towns clean, safety, and their potential energy regeneration from wastes generated, for example, the products can be used as fuel for combustion or be converted into other fuel types.

In contrast to the advantages of landfill site operations outlined above, there are major downsides associated with them, which lead to dire environmental and health repercussions. Danthurebandara et al. (2012) identified that, due to construction of landfills, the loss of habitats is demolished. This is due to the removal of topsoil, which alters their physical and chemical properties like porosity, density, strength, and water holding capacity. Moreover, there is a destruction of vegetation which could take years to be recovered.

The most important drawback of landfill operations is the formation of leachate, which is abundant in inorganic and organic chemicals, as well as heavy metals (Vithanage et al., 2017). It is caused by the chemical reaction of moisture and rainfall with garbage, resulting in a liquid that percolates into groundwater and contaminates it (Abunama et al., 2021). It also pollutes surface water and soil through runoff. In many developing countries, landfilling is the most common method of waste disposal (Tan et al., 2015), but it is often combined with inadequate leachate collection and treatment facilities, which encourages contamination (Ololade et al., 2019). In South Africa, landfilling is also a common technique of waste disposal due to the presence of available space and its costs (DEA, 2012). However, both waste generators and municipalities are unwilling to follow the regulation which puts landfilling as the last resort of disposal, as a result, landfilling becomes inevitable. The National Environmental Management: Waste Act 59 of 2008, emphasises the necessity of waste information management and trash classification before disposal (DEA, 2012). It is clear that environmental pollution (groundwater, surface water, and soil) are affected by poor infrastructural development and compliance to regulations by the parties involved in managing wastes at landfill sites.

2.2.3 Landfill classification

The waste classification system that characterised landfill operations according to the type of waste received at landfill sites was implemented by DWAF (1998).

- **Hazardous waste landfills**

Hazardous wastes can have a negative influence on the environment, even at low quantities. This is mainly due to their physical and chemical characteristics which include toxicity and corrosion. The types of hazardous wastes include: inorganic (fibrous asbestos, heavy metal sludge, acids, and alkalis), organic waste (organic chemical residues, paint and resin), oily waste from primary processing stations, wastes derived from animal and vegetable oils and products, sewage sludge, dredge spoils, soil, and building rubble. They also include wastes from hospitals (medical wastes), laboratories, and explosive wastes derived from manufacturing activities (DWAF, 1998). Hazardous landfills, therefore, are classified by the hazardous wastes they receive.

- **General waste landfills**

Unlike hazardous wastes, general wastes do not endanger human health or the environment. This is also because of their chemical and physical composition. However, they may contain certain concentrations of hazardous material spread within them, for example, domestic, commercial, medical, and agricultural waste such as insecticides. These types of wastes are disposed of at any landfill and are characterised by their ability to generate leachate (DWAF, 1998).

- **Sanitary and non-sanitary landfills**

Idowu et al. (2019) defined a sanitary landfill as a fully functional, properly lined, and well-designed infrastructure with a leachate collection system (liners and clay consolidation), as well as gas collection, cover, and a post-operation plan. These are landfills having high state-of-the-art facilities and they are also termed engineered landfills. As already stated above, landfills are predominant in developing economies that seek to minimise environmental impacts through their design and well-defined infrastructure (Gutierrez et al., 2019). Sanitary landfills are not only limited to a certain specific waste but they can accommodate various types of waste, including municipal solid waste or everyday used products such as furniture, clothing, appliances, bottles, batteries, and paint (Ozbay et al., 2021).

Sanitary landfills are not perfect in controlling contamination posed at the environment but they can minimise the impact if well managed. According to Idowu et al. (2019), engineered landfills in Ghana confront issues such as inadequate monitoring systems, clay lining, and cost-effective construction materials. Xiang et al. (2019) claimed that leaks in sanitary landfills can occur during construction and operation. Furthermore, in the Philippines, there are a total of 960 landfill sites, with 936 being sanitary and the remainder being open dumps. According to Mendoza et al. (2017), the local administration is unable to manage these landfills in a sustainable manner due to a lack of technical help and funds. In South Africa, landfills are classified into class A−D according to their design by the DEA as mandated by the National Environmental Management: Waste Act 59 of 2008, where class A is a sanitary landfill as classified by design and waste type. According to the regulation, hazardous wastes should be disposed at this type of landfill (see Figure 2.1).

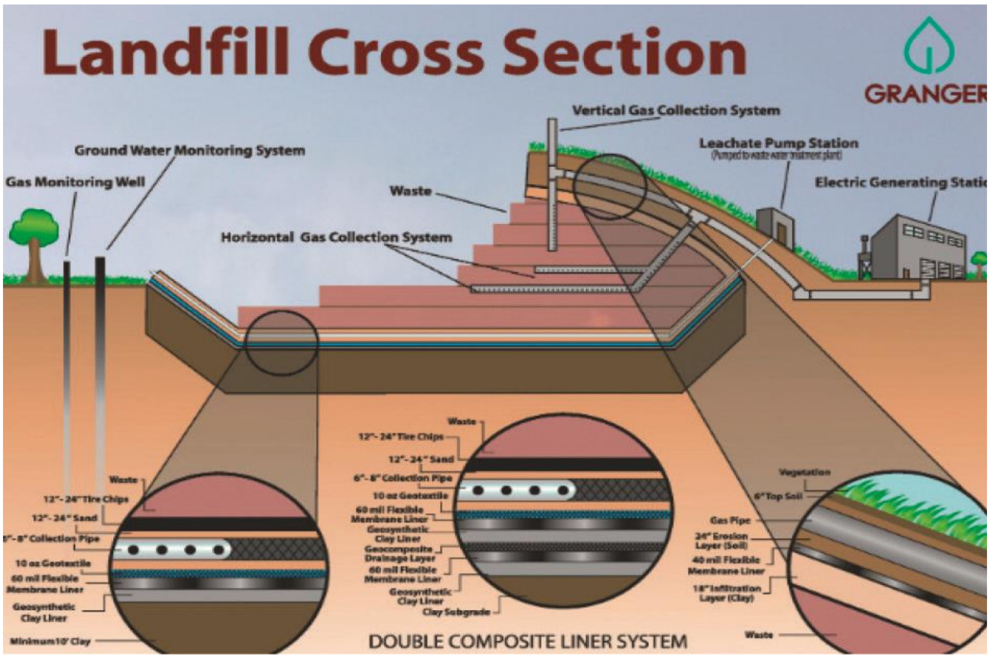


Figure 2.1 The diagram indicating a sanitary landfill *(Gulnihal et al., 2021)*

In contrast to sanitary or engineered landfills, non-engineered landfills do not meet the minimum operational requirements which are properly designed facilities. This type of a landfill is operated without considering any of the control parameters that in turn leads to severe environmental impacts (Idowu et al., 2019). The United Nations [UN] (2018) asserted that in the absence of control and regulation, open dumping often takes place, leading to leachate migration to groundwater and surface water, and uncontrolled burning which emits harmful gases into the atmosphere. This type of landfilling is common in developing countries, particularly in sub-Saharan Africa due to insufficient data, poor implementation, and understanding of the waste management systems (Dladla et al., 2016). This, therefore, asserts that more work has to be done in developing countries in terms of research.

Cases of pollution were recorded in South Africa and were triggered by the operation of non-sanitary landfill sites. For instance, Ololade et al. (2019) reported that water sources as well as soil were contaminated by the leachate in the northern landfill site in Bloemfontein. Machete (2017) also reported that due to neglect of leachate monitoring technologies, there has been decomposition, leachate, and landfill gas generation that have a severe environmental impact. The operation of non-sanitary landfills is typical in South Africa despite efforts being made to regulate and license these facilitates to meet the standards outlined in the national Gazette Notice 635 (DEA, 2018). The DEA (2018) reported that in 2017 there were about 161 hazardous landfills, with about 44% being licensed, and 990 general landfills where only 54% were licensed.

The phenomenon of the use of non-sanitary or non-engineered landfilling and associated environmental pollution still remains a huge problem, mostly in developing countries, and needs to be researched.

- **Leachate generation and contamination**

The excess of moisture at landfill sites triggers the generation of leachates. The leachate is formed as rainwater and other excess fluids in the landfill infiltrates through the waste (Figure 2.2); thus, becoming contaminated by biochemical disintegration of waste as well as dissolution and suspended matter coming from the decomposing waste (Brennan et al., 2016). The resultant liquid is highly concentrated in heavy metals, organic and inorganic compounds (Talalaj and Biedka, 2016). The leachate is mobilised to aquatic habitats via percolation at the landfill's bottom layer via the unsaturated layers until it reaches groundwater (Mishra et al, 2019). Talalaj and Biedka (2016) asserted that the leachate contamination has serious environmental problems at uncontrolled and non-engineered landfills, which is the case in developing nations. Also, heavy flooding in the wet season can favour the direct contamination of surface water by leachate runoff from the waste (Hossain et al., 2018). Several factors influence leachate migration to groundwater sources, including the composition of the leachate, the stratigraphy of the underlying soil, and the hydraulic parameters of the groundwater system (Parvin and Tareq, 2021).

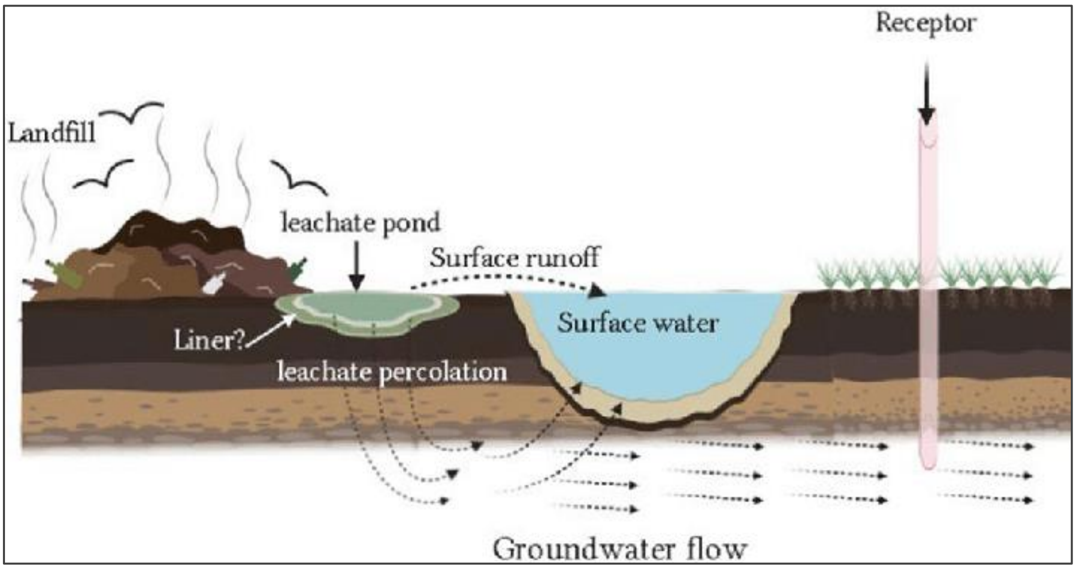


***Figure 2.2 Leachate formation and migration** (Parvin and Tareq, 2021)*

The soil surface is also polluted by the migration of leachate from the landfill sites. Ololade et al. (2019) found that heavy metals, polyaromatic hydrocarbons, and pharmaceutical products pollute soil, which is enhanced by leachate. However, the degree of contamination of groundwater, surface water, and soil by the leachate relies on the chemical composition, age, depth, accessible moisture and oxygen, meteorological condition, landfill design, and chemical and biological waste stabilisation processes within the landfill. Several researchers have studied these factors that influence the leachate composition, while others have discovered that not only leachate poses an impact but rather anthropogenic activities also directly pollute the environment.

Adhikari et al. (2014) conducted a review of the elements that influence the composition of landfill leachate. They said that waste composition activates biological activity, where organic wastes such as food, garden, agricultural, and animals might encourage biological activities to form a leachate rich in organic matter. Contrary to organic wastes, inorganic wastes such as plastics and demolition wastes, can produce a leachate.

The age of the landfill has an impact on the concentration of produced leachate. Kamaruddin et al. (2017) discovered that the duration of a landfill operation has an effect on the quality of the leachate. Mohd-Salleh et al. (2020) found out that young leachates tend to have a lower pH value compared to older ones. They indicated that during waste stabilisation phases, leachates were at acid while methanogenic phases showed low concentrations. Older leachates are much more concentrated than younger ones due to residence time in the landfill. Moisture content can influence the composition of the resultant leachate. Adhikari et al. (2014) mentioned that when there is high moisture (rainfall or excess water) in the landfill, dilution effect takes place where soluble substances, metals, microbial cells, and other compounds are expelled out of the landfill. Makhadi et al. (2020) made a similar observation during the wet season as rainfall facilitated decreased concentration of physicochemical elements in leachates.

As mentioned by Talalaj and Biedka (2016), improperly designed landfills facilitate contamination of groundwater, surface water and soil. Parvin and Tareq (2021) also asserted in their study that this toxic leachate poses risks to the environment by the use of non-engineered landfills, which is the case in developing nations.

Although many studies have been undertaken by different researchers throughout the world regarding leachate contamination and risks to both the environment and humans, there is still a lack of research in understudied areas. In South Africa, only major provinces and cities have had numerous studies conducted in them (Nyika et al., 2020), while the rest have been left untouched or have had little work done on them.

2.3 Water quality

2.3.1 Groundwater quality

Water is an essential component of human life and activities such as agriculture, industry, domestic, transportation, and health care, and it is one of the most vulnerable resources in the environment (Al-Badaii et al., 2013). The NWA states that water is the most basic requirement for life, and that no human, plant, animal, or living organism can survive without it. Groundwater is the most desired source of drinking water since it is easily accessible to various populations and is free of odour and contamination when compared to tap water (Fosso-Kankeu et al., 2019). According to Makhadi et al. (2020), groundwater is safe to drink due to its low microbial load and minimal treatment before consumption. Edokpayi et al. (2018) also stated that groundwater pollution is negligible due to its low turbidity. Groundwater is commonly used in small towns in underdeveloped countries, whereas surface water (rivers and lakes) is used in large cities, where treatment occurs first (Amano et al., 2021). Groundwater is often used as the primary source of water supply in tiny villages in South Africa’s Free State province, including Soutpan and Dewetsdorp in the Mangaung district. Groundwater is used sparingly in South Africa, with just around 9% being used, providing an opportunity to compensate for limited surface water supply of drinkable water due to ongoing droughts (Fosso-Kankeu et al., 2019).

Despite the safety of groundwater to contamination, it can be vulnerable to contamination as well. Lee et al. (2017) and Makhadi et al. (2020) have shown in their studies that several factors have a severe impact on groundwater, which include geology, climate, and anthropogenic (man-made) activities.

- **Factors affecting groundwater quality**

- ***Climate***

Vongdala et al. (2019) examined heavy metal buildup during both dry and wet seasons. Their findings revealed higher amounts of Lead (Pb) and Chromium (Cr) in groundwater during the dry season than during the wet season, implying that lowered conditions associated with evaporation increased the intensity of the produced leachate. Pb had an average value of 0,04 mg/ℓ, and Cr had 0,06 mg/ℓ, both of which were above the allowed limits of the World Health Organization (WHO) and the Laos Agreement on the National Standards (Vongdala et al., 2019). During the wet season, however, the diluting effect produced low-strength leachate. Makhadi et al. (2020) made comparable observations on the physicochemical properties of groundwater and surface water in both seasons.

Contrary to the findings above, more precipitation during the rainy season can lead to groundwater contamination. Palomeras et al. (2021) found that the wet season contaminated well water and groundwater. The Total Dissolved Solids (TDS) concentration was 984 mg/ℓ, which surpassed the WHO standards and the national criteria of the Philippine National Standards for Drinking Water (Republic of the Philippines, 2017) of 500 mg/ℓ and 600 mg/ℓ, respectively. The turbidity was 13 Nephelometric Turbidity Units (NTU), which was greater than the standards of the WHO and the Philippines of 4 NTU and 5 NTU, respectively. According to Mendieta-Mendoza et al. (2021), TDS is a water quality measure that demonstrates mineral deposition and that the groundwater is rich in suspended particles. Both turbidity and TDS account for the murky colour of the water due to particulates introduced by runoff of rainwater into groundwater bodies.

- ***Geology and geohydrology of an area***

The underlying geology or soil stratigraphy can either minimise or maximise the degree of contamination of groundwater sources at landfills. A groundwater contamination by leachate study was conducted in Lagos, Nigeria. Aderemi et al. (2011) found that the leachates showed higher concentrations of physicochemical parameters than groundwater. For instance, pH was 8,1 for leachates as opposed to 5,0 for groundwater, and heavy metals Pb and Cadmium (Cd) were high in leachates, whereas they were not detected in groundwater samples. They observed that the effect of soil stratigraphy minimised contamination as the area has abundant clay layers that adsorb metals due to their affinity. Although stratigraphy can help to reduce pollution, geological characteristics, such as faults and joints/fractures, can also increase the risk. A similar study was carried out in the Eastern Cape province of South Africa, where local geology consisted of clay interlayered with sandstone, but the presence of fractures within the rock layers brought about contamination (Mepaiyeda et al., 2020). This is because the fractured rock serves to provide pathways for the movement of the contaminant into groundwater.

The presence of permeable rock layers also facilitates contamination of groundwater by the leachate percolation. Golden and Inichinbia (2020) discovered contamination associated with increasing resistivity values where the maximum value was 248 RΩ characterised by permeable sandy layers with variable grain sizes, while the minimum value was 47 and 78, having topsoil and laterite clay. They concluded that resistivity increased with increasing grain sizes from fine to coarse-grained strata that is defined by clay to coarse, weak sandstone.

Furthermore, as explored by Santhosh and Babu (2018), the type and depth of an aquifer are natural characteristics that determine groundwater potential for pollution in an area. Confined aquifers will experience less pollution by the landfill leachate than unconfined ones because of the presence of an upper clay layer acting as a protective barrier against the action of pollutants infiltrating into the water table. The opposite is the case with unconfined aquifers which experience an ease of pollution due to the absence of a protective cap overlying it (Rukmana et al., 2019). Again, aquifer depth from the surface will impact the pollution of the groundwater. Deeper aquifers will have less pollution by the action of the leachate than shallow ones, and the reason is that with deeper depth contaminants tend to travel larger distances and for longer periods of time (Nnadozie et al., 2019). This then causes the surrounding rock layers and stratigraphy to react with the pollutant, leading to attenuation through adsorption caused by ions from the rock layers. The opposite is true regarding shallow aquifers which allow for greater pollution due to short travel and less reactive times between pollutants and the strata.

- ***Anthropogenic activities***

Human activities rely on water for their successful execution and sustainability; however, the very same activities can in turn pose deleterious effects on the same water resources in a direct or indirect way. These human activities include farming, industrial activities, and human settlements due to population growth (Ololade et al., 2019). Groundwater is used for agricultural activities, especially in places that have dry climates, and therefore its quality is of the utmost importance for local and national governments (Asadi et al., 2019). Several studies have reported contamination of groundwater sources by agricultural wastes being dumped on landfill sites. The groundwater quality was reportedly deteriorated by the presence of Cd metal by the agricultural fields that were located next to the landfill (Przydatek and Kanownik, 2019). Although it was in very small concentrations it rendered the water unsuitable for consumption. The report on the Integrated Waste Management Plan [IWMP] (IWMP, 2011) of the Mangaung Metropolitan Municipality (MMM) shows that extensive farming activities occur in the region, which produces hazardous wastes in the form of pesticide containers and other chemicals. Furthermore, Ololade et al. (2019) have recorded high concentrations of Bromide (Br), that were attributed to human activities, including agriculture. Industrial effluents also have their fair share of contamination of groundwater sources that they come into contact with. Heavy metals such Cd, Copper (Cu), and Zinc (Zn) have been increasingly used and produced in industrial sites over the years (Naveen et al., 2018). By their nature, industries release very hazardous types of waste due to commercial, agricultural and mining activities (DWAF, 1998). Moreover, many industries dispose of their waste in unlined soil pits and landfills, where they easily migrate to groundwater. As a result, heavy metals, for example, Cd, Cu, Arsenic (As), Pb,Cr, Ammonium (NH_4^+), and Nitrate (NO_3^-) pollute groundwater and endanger human health through consumption (Han et al., 2013).

Economic and industrial development lead to urbanisation, which exerts pressure on solid waste management (Kaza et al., 2018). With the rise in population, more waste is generated as indicated at the beginning of this discussion, and therefore finds its way to municipal landfills. Many cases of pollution directly at landfills have been reported around the world and locally, however, indirect pollution to groundwater resources also has an impact. Gqomfa et al. (2022) investigated the effects of informal settlements on the quality of rivers. They observed that an increase in human informal settlements as the population increases has had an impact of the quality of water. For instance, due to disposal of waste in wetlands and drains, the contaminants end up in rivers, which in turn has polluted groundwater through the recharge process.

2.3.2 Surface water quality

For many decades, human growth has been dependent on surface water, which plays an important role in a country's environmental, social, and economic development. Surface water sources, such as rivers, account for around 0,006% of all fresh water sources on the planet, posing a significant concern. Surface water sources such as rivers, lakes, and dams provide around one-third of all drinking water. Surface water is primarily used as a source of drinkable water for household, agricultural, industrial, and recreational applications worldwide. Pollution poses the greatest threat to this valuable resource, particularly in emerging countries like India, where about 100 million people live near river banks (Shio et al., 2015). In South Africa, many people rely on untreated water from surface water sources such as lakes and rivers for survival. This is due to inadequate sanitation and water treatment technologies, which require people to discharge both solid waste and waste water into these resources, polluting them (Edokpayi et al., 2017).

The quality of water is defined by its physical, chemical, and biological qualities, which affect its suitability for diverse applications. However, its ease of access has led in the degradation of its quality and availability over time (Edokpayi et al., 2017). While the natural system has the ability to purify itself via processes such as biodegradation and dilution, human activities have accounted for their constant decline in quality, leading to greater purification costs and sustainability. Flooding, drought, weathering, and erosion are examples of natural phenomena that can have an impact on surface water quality (Delpla et al., 2009; Seiyaboh and Izah, 2017). This decrease in water can lead to food insecurity due to poor crop production through polluted irrigation water, poor health of humans through consumption of microbiologically polluted water, as well as the usage of contaminated groundwater via the recharge process (Edokpayi et al., 2017).

- **Factors affecting surface water quality**

- ***Natural factors***

Flooding and rising temperatures are two ways that climate change may affect surface water quality. Delpla et al. (2009) investigated the effects of climate change on surface water quality. They stated that climate change would frequently result in severe rainfall, which will carry suspended solids and other debris into streams, rivers, and other bodies of water, polluting them. Again, rainfall has the potential to dilute or enhance pollutant concentrations in streams. Mekonnen et al. (2020) underlined that excessive runoff increases the rate of water transfer into surface waters, resulting in pollution. This was supported by the increased values of NO_3^- in surface water bodies near a farming area in Tepi Town, Ethiopia. In addition, during high rainfall events, erosion and deposition of the surrounding soil, organic matter and other substances occurs, as evidenced by heightened total suspended solids and turbidity in water (Reggam et al., 2017).

Variable temperatures also play a significant role in the quality of surface water. Temperature affects all the physicochemical parameters because chemical reactions take place under certain temperatures. Therefore, processes related to water change are tied to an increase in temperature and they are: evaporation, degradation, solubilisation, and dissolution (Delpla et al., 2009). This then leads to an increase or decrease in dissolved substances in water. In addition, several researches have been conducted on the impact of sampling season between hot and cold seasons, and they all concluded that during the hot season, which is associated with higher atmospheric temperatures, there were more elevated concentrations of physicochemical parameters in leachates, surface water and groundwater than in the cold season (Angaye and Mieyepa, 2015; Ololade et al., 2019; Seiyaboh and Izah, 2017).

- ***Anthropogenic factors***

Anthropogenic activities degrade surface water in higher quantities than natural processes will ever do. There are two ways for pollution to occur: directly and indirectly. For the purposes of this study, indirect contamination is emphasised and concentrated on the most. As previously noted in chapter 2, the effects of human activities on groundwater, namely population growth, industrialisation, and agriculture, are likewise polluting drivers in surface waterways. Alam and Qiao (2020) mentioned in their research that in Bangladesh, waste generation with rapid population, coupled with improper solid waste management, led to pollution of surface water bodies adjacent to landfill sites. The findings also indicated that waste leachate contamination in a city is directly related to the population size. This was corroborated by the results on physicochemical (inorganic) parameters and heavy metal concentrations that were enhanced in surface waters in Bangladesh cities, with one city having a bigger population than the other. The enrichment exceeded the WHO and the Department of Environment in Bangladesh as a result of population growth and landfilling (Parvin and Tareq, 2021).

In addition, the distance of landfill sites as planned by governments and authorities relative to surface water bodies plays a major role in their quality. Many studies have discovered that surface water bodies situated closer to landfill sites have a greater chance of leachate pollution than those located farther away (Mekonnen et al., 2020; Olukanni et al., 2017). Agricultural and industrial wastes are also dumped at landfills, polluting surface water bodies. The result of this dumping has introduced hazardous wastes, including heavy metals, into the water and soil environment (Hailemariam and Ajene, 2014). Therefore, high concentrations of Pb, Cu, Nickel (Ni), Manganese (Mn), Electrical Conductivity (EC), pH, and Cd were enriched in leachates and surface water bodies around landfills (Mekonnen et al., 2020).

2.3.3 Water quality parameters

Water quality parameters measure the acceptability of water for drinking and irrigation in most circumstances. They consist of physical and chemical, often referred to physicochemical parameters, as well as biological parameters. They are required to be within certain limits for the water to be deemed fit for the above-mentioned purposes, but when they do not meet up to the defined limits, then the water in not suitable for such uses. The water quality parameters described below are prescribed and necessary in the water management process for assessing water quality. This study will exclusively address the physicochemical parameters.

- **pH**

The pH test determines the acidity and alkalinity of water. The optimal pH range for biological life to exist is between 6 and 9, although pH levels outside of this range have a negative impact on biological and human health. The breakdown stages in the municipal solid waste landfill influence the pH at which the leachate is concentrated. For example, in the breakdown of organic matter, carbon dioxide under pressure alters the pH of the leachate. In addition, dissolved pollutants and gases in the waste cause the natural pH of water to alter from acidic to alkaline. Water with a pH of less than 7 is soft due to organic acids, carbonic acids, and fulvic acids (Mahapatra et al., 2011). However, pH levels above 7 result in increased dissolved chemicals for plants but not for drinking. PH is an excellent measure of landfill maturity, with alkaline nature indicating the mature stage and acid phase indicating the young age of the dump (Naveen et al., 2016).

- **Total dissolved solids**

TDS are made up of mobile charged ions such as minerals, salts, and metals dissolved in water (Seiyaboh and Izah, 2017). They are classified as inorganic salts, which include Magnesium (Mg), Calcium (Ca), Potassium (K), Sodium (Na), and Chloride (Cl). High TDS concentrations reduce the clarity of water, impede photosynthesis, and increase water temperature. This results in the death of aquatic species by algae and bacteria (Naveen et al., 2016). Furthermore, during periods of high rainfall, TDS increases because there is more runoff and the sediments are picked up by rainwater to deposit them into streams. Moreover, during dry periods, soil erosion can lead to higher TDS in water.

- **Electrical conductivity**

The presence of dissolved organic and inorganic components in a solution influences EC, just as it does TDS. EC indicates the mineral concentration and salinity of the leachate (Naveen et al., 2016). When water contains a high concentration of mineral salts and dissolved ions, its conductivity increases (Amangabara and Ejenma, 2012).

- **Chemical oxygen demand**

Chemical Oxygen Demand (COD) is the volume of Dissolved Oxygen (DO) required to oxidise and stabilise the organic and inorganic composition of water. COD in polluted water exceeds the Biological Oxygen Demand (BOD) due to oxidation processes (Seiyaboh and Izah, 2017). As such, oxidation processes take up available oxygen needed for aquatic species like fish to survive, and this leads to their decay. COD increases during the waste stabilisation phase inside the landfill in the acid formation phase where pH decreases sharply with the degradation of nutrients (Adhikari et al., 2014).

- **Total organic carbon**

The assessment of soil and water pollution, as well as the organic matter in sediment, is reflected in the Total Organic Carbon (TOC). The behaviour of organic carbon is reflected in the TOC, which is a measure of sediment organic matter and water and soil contamination (He et al., 2016). As a result of the influence of natural process (climate, temperature) as well as anthropogenic activities (agricultural, industrial), even water quality parameters such as TOC have been widely used in conjunction with COD and BOD to better understand the water quality assessment. It is also an indicator of non-biodegradable organic matter from non-point sources (Lee et al., 2020).

- **Major cations**

The main cations found in leachates are Ca, Mg, K and Na. They give information about the sustainability and productivity of water (Seiyaboh and Izah, 2017). Furthermore, the kind of waste deposited and the waste stabilisation phase within the landfill determine the concentrations of cations that are transferred from the waste into the leachate by mass transfer mechanisms.

- **Major anions**

Constituents such as Cl, Sulphate (SO_4), and NO_3^- make up the anion as the parameters to measure water quality. Though they occur in relatively low conce ntration water bodies, they can be elevated through human activities. For instance, waste deposited at landfill sites is reported to have increased the concentrations of all the anions in the Mallipura landfill in India. In this landfill site, Cl had the highest concentrations attributed to have been caused by the leachate pollution from the waste that came from the industrial, domestic, septic tanks, and fertilisers. SO_4 originates from the decomposition of organic waste in the landfill, whereas NO_3^- also originates from agricultural effluents containing fertilisers (Naveen et al., 2016).

- **Heavy metals**

A metallic element is considered a metal if its specific gravity is equal or greater than 5 cm^3. They are grouped according to their significance, having an important role in the human body. These include Zn, Iron (Fe), Cu, and Mn. Conversely, harmful metals like Cd, Pb, and Mercury (Hg) have a major negative impact on both human health and the environment. The impacts of heavy metal pollution at landfills have been the subject of numerous research (Ololade et al., 2019; Tsarpali et al., 2012). Heavy metals are very dangerous even at trace amounts when they get into contact with the environment, and unlike many other organic pollutants, very hazardous metals cannot be destroyed nor can they be chemically or biologically degraded, therefore, they remain in the environment for long periods (Qian et al., 2021).

2.3.4 Water quality guidelines/standards

The United States Environmental Protection Agency [USEPA] (USEPA, 2012) stated that water quality standards are laws and rules designed to enhance the quality of water to safeguard the public's health and well-being. Restoring and preserving the chemical, physical, and biological integrity of a nation's water is the goal of water quality standards. USEPA (2012) further outlined that the standards are classed into three categories, mainly: indented use or uses of water or a water body; water quality criteria purposed to protect the use of such water or water body; and an anti-degradation policy.

- **World Health Organization guidelines for drinking water**

The international water quality standard used in nations worldwide is set by the WHO. The organisation's purpose for drinking-water quality is mainly to protect health by managing the risk from consumption that can pose intensive hazards to drinking water (WHO, 2011). The WHO (2011) stated that having access to drinking water of higher quality is essential for positive health effects. Safe drinking water is essential for household tasks such as drinking, cleaning, and cooking, but it might not be applicable for other uses such as leisure. An emphasis in the guidelines is the means to develop and implement the risk management strategies to ensure safe drinking water by managing the hazardous constituents of water. The guidelines, hence, define minimum requirements for safe limits for water usage by giving values for parameters of water quality (refer to Table 2.1). Regarding Table 2.1, it should be noted that the WHO does not give certain limits to all the parameters, for instance, TOC, COD and Cobalt (Co); therefore, countries should also play their role as there is no single approach that is used worldwide. This serves to mention that each country should formulate its own safety guidelines that fits its national needs.

- **South African National Standards for drinking water**

South African National Standards (SANS) sets the national drinking water quality standards for South Africa. In accordance with the Standards Act 24 of 1945, the South African Bureau of Standards [SABS] (2015) established SANS to promote and preserve the quality of the country's commodities. SANS, like the WHO, has set minimum standards for the microbiological, physical, and biological properties of water (SANS, 2015). If the water is determined to be within the SANS-specified limitations, it is deemed safe for drinking and will not have any negative health impacts over the course of a lifetime of usage. SANS lists the water characteristics that must be examined in addition to providing minimum values.

- **Department of Water Affairs and Forestry guidelines**

South Africa's water resources are under the DWAF (1996) jurisdiction. It uses four categories – recreational, industrial, residential, and agricultural use – to assess water consumption with the goal of maintaining its fitness in a sustainable way. DWAF provided a methodical approach to assessing the suitability of water use for a given purpose. To determine this fitness, one is required to outline the specific purpose for water use by determining the water quality requirements for that use by gathering data about constituents that measure water quality, and to find ways in which to reduce the water quality impact should the water be deduced to be contaminated. Water quality ranges are also given by DWAF in measuring water quality, which follow this order: ideal (100% fit for use), acceptable, tolerable, unacceptable, and completely unfit for usage. The use of these water quality guidelines are accessible to stakeholders, decision-makers, water quality managers, and other interested parties, including members of the general public.

Table 2.1 National standards guidelines

Parameters	WHO (2015)	SANS 241 (2015)	DWAF (1996)
pH	6,5−8,5	5,00−9,70	6,50−8,40
TDS	500,00	1 200,00	−
EC	1 500,00	1 700,00	<40,00
COD	−	−	−
TOC	−	≤10,00	−
Ca	75,00	300,00	−
Mg	30,00	100,00	−
Na	200,00	≤200,00	0,00−70,00
K	300,00	100,00	−
HCO_3	−	−	−
SO_4	500,00	≤500,00	−
Cl	250,00	≤300,00	0,00−105,00
Mn	−	≤0,40	≤10,00
Cr	50,00	3,00	0,10
Cd	3,00	10,00	0,003
Co	−	50,00	0,05
Fe	−	300,0	5,00
Pb	−	0,01	1,00
Zn	−	0,50	0,10
As	10,00	0,01	0,10
V	−	4,000	0,20
Cu	−	2,000	0,20
Se	40,00	0,010	0,02

The mean values for COD, TDS and TOC are given in mg/ℓ, and pH is in pH units. Total coliform and *E. coli* are expressed in colony forming units per 100mℓ. The standards for water quality are SANS 241 (2015), WHO (2015) for drinking purposes and DWAF for irrigation purposes.
Source: WHO (2015), SANS 241 (2015), DWAF (1996).

2.4 Pollution indices

2.4.1 Basic concepts, history and application of water quality indices

A deeper comprehension of the fundamental ideas and principles of water is necessary in order to use the Water Quality Indices (WQI), which are techniques and instruments used to assess the general state of water (Bharti and Katyal, 2011). The WQI is a well-known technique for reflecting the quality of drinking water. It provides a numerical value in response to changes in the properties of water (Ravikumar et al., 2013). According to Namugize and Jewitt (2018), the WQI is vital since it converts complex data of water quality parameters that are difficult to interpret, into simple mathematical values by converting concentrations of pollutants into a number. As a result, they can be used by researchers, decision-makers, the public, and the managers in programme design and assigning policy details (Allam et al., 2015). According to Namugize and Jewitt (2018), water quality indices are essential for determining which water contamination issues need particular focus and attention as well as for evaluating how well water quality monitoring systems are working. Even while they are crucial for improving the communication of water quality, they have drawbacks of their own, which has caused numerous academics and governments worldwide to develop alternative water quality indices. They include the Canadian Council of Ministers of the Environment (CCME) WQI, the Vaal WQI, and the Malaysia WQI (Banda, 2015; CCME, 2001).

The usage of water quality indices began in the middle of the 20th century when Horton initially figured out how to classify water based on its level of purity (Bharti and Katyal, 2011). Using pH, DO, Cl, conductivity, alkalinity, and coliforms, Horton (1965) chose the ten most frequently chosen and measured water quality characteristics for his WQI. The WQI range was between 1 and 4, and he determined the index's score using the linear arithmetic tool. The function involved multiplying the two coefficients (M1 and M2), which represented the temperature and pollution reflections, by the sum of the weighted sub-indices divided by the total sum of the weights. Horton's work, nonetheless, did not end there, as it was adopted by many pioneers who have further formulated different WQIs that have been widely accepted by various organisations for their water supply and control of pollution (Bharti and Katyal, 2011). The methodology upon the development of the indices formed the original concept; however, it remained the same, which is mainly the selection of parameters, deriving the formula, followed by index scale, and the application of guidelines/boundaries.

2.4.2 Phases of development of water quality indices

The WQI as described by Chidiac et al. (2023) are graphically shown in Figure 2.3 and were developed in the following phases:

- Water quality measurement parameter selection: This encompasses a broad variety of water quality metrics that have a direct impact on the water quality from different study locations. Physical characteristics, health indicators, amounts of oxygen, eutrophication, and dissolved phases can all be used to determine the water quality.

- Raw data parameter conversion, which converts parameters from their unit measurements into sub-indices and a common scale (Paun et al., 2016).
- Provision of weights to parameters in accordance with their importance and level of contribution to water quality. International and national standards are often utilised.
- The final WQI, which is the sum of the ratings and weights, is obtained through sub-index aggregation.

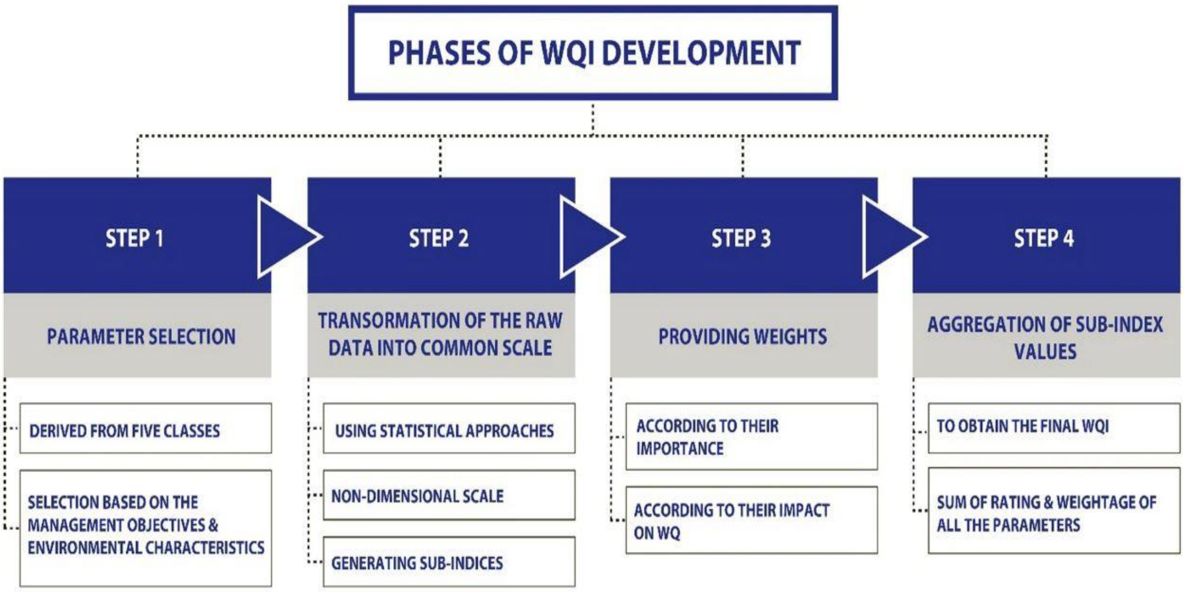


Figure 2.3 Phases of WQI development *(Chidiac et al., 2023)*

2.4.3 Categories of water quality indices

Poonam et al. (2015) and Sobhani (2003) divided water quality indices into four major categories, with the last one concentrating on planning rather than necessarily the manual application of the index, while the first three take application into account. The following is a brief classification of the indices.

- **Public indices:** They are characterised by the general use of consumption purpose of the water; therefore, they are not limited to specific requirements. An example is the Malaysian WQI (Horton, 1965). Moreover, their application is relevant in any field of water quality assessment, including industrial, agricultural, recreation, and solid waste management.
- **Specific indices:** In this category, as it is self-explanatory, the indices are developed for a particular purpose of water consumption (drinking, irrigation,

and recreation). In this instance, the examples include the Vaal WQI by Banda (2015) for evaluating the raw surface water quality for the treatment of potable water standards and the Landfill Water Pollution Index (LWPI) for evaluating groundwater pollution caused by landfilling (Talalaj and Biedka, 2016).

- **Statistical indices:** In this category, only statistical methods are applied and no personal opinions are implicated. The analysis is, therefore, objective, based on the available data given and interpreted (Bharti and Katyal, 2011).
- **Planning indices:** This category is composed of tools and instruments that has been intentionally designed to aid policymakers and managers to make better decisions on the water quality. They act as guidelines for decision-making. The example includes routine stream monitoring index (Banda and Kumarasamy, 2020).

2.4.4 Description of some common and major water quality indices methods

- **National Sanitation Foundation water quality index**

The National Sanitation Foundation (NSF) in the United States created the NSF WQI in 1970 (Hamlat et al., 2017). Numerous researchers have used this WQI approach in a variety of water bodies. As a member of the public indices group, it is an acronym for the general WQI and does not account for water use or capacity. Additionally, this approach uses nine parameters for analysis, which include temperature, turbidity, pH, faecal coliforms, DO, BOD, NO_3^-, total phosphate (PO_4^{3-}), and total solids. Because certain parameters were more important than others in determining the NSF WQI, a mean weighting of each parameter was done with the input of specialists that were based on environmental importance and water body use who made a total of weight equalling 1 (Chidiac et al., 2023). The mathematical formula for computing the NSF WQI is given by the following equation:

$$NSFWQI = \sum_{i=1}^{n} QiWi \qquad \textit{Equation 2.1}$$

Where the sub-index i^{th} water quality parameter is denoted by Qi. The number of water quality parameters is n, and Wi is the weight associated with the i^{th} parameter.

This approach uses a scale from 0 to 100, where 100 denotes ideal water quality conditions and 0 denotes inappropriate water use that requires additional treatment.

- **Canadian Council of Ministers of the Environment Water Quality Index**

The CCME WQI provides an evaluation of the WQI to determine whether water bodies are suitable for supporting life in Canadian regions. Information on the water quality for public areas and management is provided by this WQI (Paun et al., 2016). Because it simplifies extremely complex and technical information, this WQI approach can be used by a variety of national water authorities. The application of this technique evaluates the multi-variable data and contrasts it with the user's data, as claimed by Tirkey et al. (2015). It does not recommend the specific parameters to be used but it only requires four of them, which can differ from one place to another. The method also has a class range between 0 and 100, whereby 100 signifies the method to have attained a benchmark. It utilises an automated spreadsheet that has numerical calculations in the following formula:

$$CCME\ WQI = 100 - (\sqrt{F1^2} + \sqrt{F2^2} + \sqrt{F3^2} \div 1{,}732$$ *Equation 2.2*

Where,

F1 = $[\frac{Number\ of\ failed\ tests}{Total\ number\ of\ variables}] \times 100$

F2 = $[\frac{Number\ of\ failed\ tests}{Total\ number\ of\ tests}] \times 100$

F3 = $[\frac{nse}{0{,}01(nse)+0{,}01}]$, where $nse = \frac{\sum_{i=1}^{n} excursion j}{total\ number\ of\ test}] - 1$

Although the CCME WQI has several uses, including for surface water bodies such as rivers, lakes, and streams, it has drawbacks related to the fact that it requires a special way to combine several variables to find missing data. Additionally, the WQI lacks biological data, its results are reliant on variables and failed samples, and it cannot be used in many ecosystems (Kumar et al., 2024).

- **Dinius water quality index**

Water pollution control methods were evaluated using the Dinius WQI, which employed simple equations for sub-index computations (Aljanabi et al., 2021). The Dinius WQI is used to assess the degree of contamination in freshwater sources, including rivers and streams. The contaminants included in the index determination, as well as the relationship between the concentration of the parameters and the index, was undertaken using the Delphi review, which involves a panel of experts (Kumar et al., 2024). The Delphi questionnaire included water usage categories for agriculture, industry, public water supply, recreation, fish, and shellfish. The limitations of the Dinius method are that it does not give consideration to possible contaminants in aquatic systems and does not provide information regarding changes in water quality. The index had 12 parameters that were selected that entailed conductivity, pH, DO, colour, temperature, nitrate, BOD, alkalinity, *E. coli*. Hardness and coliforms were calculated using the following function formula (Dinius, 1987):

$$IWQ = \sum_{i=1}^{n} .Iiwi \qquad \textit{Equation 2.3}$$

Wi is the unit weight of the contamination parameter with a number between 0 and 1, and *Ii* is the sub-index contaminant parameter with a number between 0 and 100, $\sum_{i=1}^{n}.$ is equal to 1, while *n* represents the number of pollutant variables.

- **Nemerow pollution index**

Introduced by Nemerow and Sumitomo (1970), the Nemerow Pollution Index (NPI) is a rapid technique for determining the WQI of water by choosing particular characteristics that contribute to pollution (Dawood, 2017). Its quickness of use is its most advantageous ability as it determines the WQI by using the concentration of parameters and their standard limits, which brings understanding between them. A contaminant that needs to be pretreated before use is present when the calculated value is greater than 1, The NPI is computed using the equation:

$$NPI = \frac{\mathrm{Ci}}{\mathrm{Li}} \qquad \textit{Equation 2.4}$$

Where Li is the permitted limit of the ith parameter and Ci is the measured value of the ith parameter.

- **Scottish Research Development Department Index**

In 1970, the Scottish government created the Scottish Research Development Department (SRDD) index to assess the quality of surface water. This model is advantageous in the way that it allows flexibility and is convenient for regional use (Uddin et al., 2021). It also uses the Delphi model in its selection of paramters which include four groups: physical group (suspended solids, temperature, and conductivity), chemical group (saline ammonia, pH and DO, organic group (phosphate, nitrogen and total oxide), and microbiological group (*Escherichia coli [E. coli]*). According to Kumar et al. (2022), the SRDD model gives an overall WQI for a water body. However, its limitations involve a lack of temporal resolution, non-consideration of ecological effect, and lack of considering of toxic substances. The formula for SRDD is:

$$SRDD - WQI = \frac{1}{100}(\sum_{i=1}^{n} SiWi)^2$$

Equation 2.5

The calculated SRDD index is classified into a seven-class range from piggery waste (0−19), several polluted (20−29), polluted (30−39), tolerable (40−69), good without treatment (70−79), good (80−89), and clean (90−100).

- **Malaysian water quality index**

The Malaysian WQI was created in 1974 by the Malaysian Department of Environment to assess the quality of surface water in accordance with national requirements. BOD, COD, ammoniacal nitrogen, pH, suspended solids, and DO were among the six factors (Uddin et al., 2021).

As previously stated, the Malaysian WQI is applied in rivers. Its limitations are that it only focuses on the chemical and physical parameters in the WQI without considering the biological and ecological factors, as well as various water uses like recreation, irrigation, and drinking (Kumar et al., 2024). Furthermore, its weighted method is not transparent, which largely prohibits its wide applicability. In the determination of the Malaysian WQI, the WQI score was computed by adding the sub-index values of the parameters in the equation below:

$$\begin{aligned} WQI = {} & 0{,}22 \times SI_{DO} + 0{,}19 \times SI_{BOD} + 0{,}16 \times SI_{COD} + 0{,}15 \times SI_{AN} \\ & + 0{,}16 \times SI_{SS} + 0{,}12 \times SI_{pH} \end{aligned}$$

Equation 2.6

Where 0,22 represents the weight to DO, 0,19 represents the weight to BOD, 0,16 represents the weight to COD and suspended solids, 0,15 represents the weight to NH_3-N, and 0,12 represents the weight to pH.

- **Bascaron water quality index**

The Bascaron WQI was invented in 1979 by Bascaron in assessing the water quality in relation to the Spanish national water quality standards (Sun et al., 2016). This method gives a benefit of improving the WQI in connection with the statistical methods like variable reduction and provides a vast range of water quality items. The more parameters that can be utilised in evaluating the index, the more precision can be achieved (Kumar et al., 2024). This index was used by different countries for assessment of lakes but gave good results when used in conjunction with other models. Its limitations are that it needs various factors for calculation that may lead to a rise in interference, resulting in low accuracy. It also glides on subjectivity in weightings allocated to various parameters and that can influence overall results. The Bascaron WQI is calculated using the following formula:

$$Bascaron\ WQI = \sum_{i=1}^{n} . C_i \times P_i \div \sum_{i=1}^{n} Pi \qquad \textit{Equation 2.7}$$

Where P_i is the relative weight for each individual water quality item, C_i is the water quality that is computed as a discrete value in the standardised value, and n is the calculated water quality item.

- **Horton water quality index**

The Horton WQI was developed in 1965 by Horton in the United States. It makes use of ten parameters of which two are temperature and the pollution level classified as adjustment parameters (Fortes et al., 2023). The other seven parameters include pH, dissolved oxygen, conductivity, alkalinity, chlorides, sewage, and treatment percentage. To calculate the Horton WQI, two methods are followed which entails weight allocation based on selected parameters and rating, as well as data aggregation. It ranges from 0 that represents high quality and 100 for low water quality. It is computed using the following formula:

$$WQI = \frac{\sum_{i=1}^{n} WiIi}{\sum_{i=1}^{n} Wi} \times M_1 M_2 \qquad \textit{Equation 2.8}$$

Where I_i is the sub-indices, W_i is the sum of weight, M_1 stands for temperature coefficients, and M_2 is coefficient for obvious pollution.

2.5 Hydrochemistry

2.5.1 Hydrochemical characterisation

The surface geology determines the quality of groundwater, the bedrock nature, precipitation, anthropogenic or human activities and climate, as mentioned in previous sections. However, by introducing pollutants into the contamination system, these effects can potentially lead to groundwater pollution. However, in the subsurface environment, additional geochemical reactions result in the actual alteration of the chemistry of groundwater sources as they flow from recharge to discharge, where they are influenced by various biological processes, weathering, ion exchange, and mineral dissolution (Sakram et al., 2013). The concept hydrochemical facies is used to delineate or display the chemical nature of water in the hydrological system by reflecting hydrochemical processes that are difficult to comprehend and occur in the subsurface environment between groundwater and lithology (Ravikumar et al., 2015).

The geochemical character of groundwater can tell a lot about its evolution which contributes to it. Realising the importance of the hydrochemistry of groundwater, researchers have detailed groundwater deterioration and its geochemical evolution in their studies (Acharya et al., 2018; Makhadi et al., 2020; Ololade et al., 2019). Among other tools, Piper, Durov, and sodium danger diagrams are used to graphically describe hydrochemical evaluation of geochemical data to determine the acceptability of groundwater for drinking and irrigation. The following sections will explain the use of each method.

- **Piper trilinear diagram**

By looking at the relationship between the sorts of rocks and the composition of water, Piper (1944) discovered how to utilise a Piper diagram to assess the growth of water. Additionally, it discloses the geochemical mechanisms influencing the composition of groundwater and, consequently, its quality (Ravikumar et al., 2015). The use of this method can also help to classify the type of water that has been produced by such water−rock interactions. Originally, natural waters are composed of less dissolved constituents, which are cations and anions that are in chemical equilibrium with each other. Moreover, natural waters also contain constituents such as Aluminium (Al), Fe and Silicon (Si) in their oxide forms – not in equilibrium with other ions (Piper, 1944).

Furthermore, natural waters also contains two alkaline rare earth elements (Ca and Mg), and an alkali (Na) but with less K. Cations and anions are in chemical equilibrium in natural waters, but when there are hydrochemical processes and other factors, the state of equilibrium can change to result in waters that have different compositions, types, and hence quality. As a result, the use of a Piper diagram is designed to outline these changes.

According to Hosseinifard and Aminiyan (2015), a Piper diagram is made up of three fields: two trilinear/triangular diagrams and a diamond-shaped diagram in the middle. On the left side of the trilinear diagrams are cations, and on the right side are anions. The main separations are Na^+ plus K^+, Ca_2^+, and Mg^{2+} ions, whereas anions are also grouped into major groups which are: SO_4^{2-},Cl^-, Bicarbonate ion (HCO_3^-) plus Carbonate ion (CO_3^{2-}). Upon plotting of these constituents in their percentages on the trilinear diagrams for both cations and anions, they are then projected into the diamond field to give the type of water that is being investigated. Figure 2.4 shows the simplified Piper diagram chart with all its water classification fields and dominant cations, anions and mixed water types.

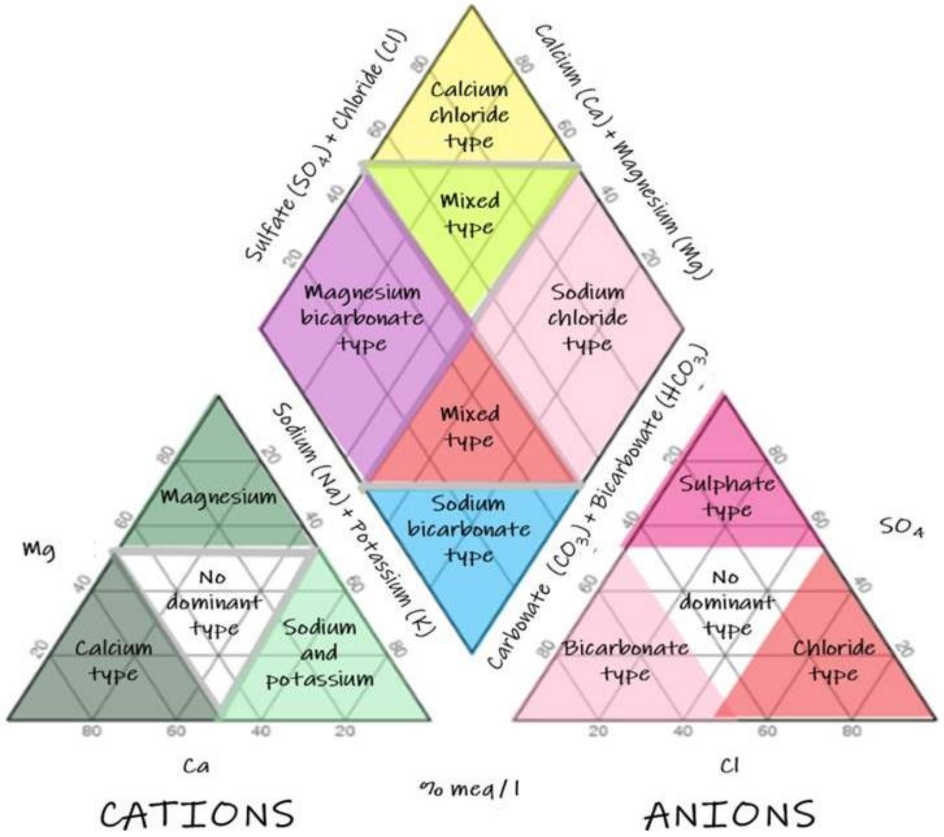


Figure 2.4 Trilinear Piper diagram *(Ogbozige and Toko, 2020)*

- **Salinity hazard (total soluble salt) diagram**

Irrigation water is used to supplement the natural precipitation by rainfall or for crop protection against wilting and freezing (USEPA, 2005). Therefore, the inefficient supply of such water creates water quality problems. Since groundwater is also used for irrigation for crops in many countries of the world, its irrigation quality is of the utmost importance for such use, and as a result, the need to evaluate its quality for irrigation is underpinned by the alkalinity or Na hazard concept. Alkalinity hazards or Na content are other ways to express the Sodium Sorption Ratio (SAR) (Srinivasamoorthy et al., 2014). High salt concentrations in irrigation water impair the permeability and water-infiltration capability of the soil. The problems of infiltration occur when water does not seep rapidly into the soil structure down into the root area as it normally should in the irrigation cycle (Ogunfowokan and Obisanya, 2013).

EC is used to measure the salinity of water. Drought results from the inability of the plant to get necessary ions from the soil because of the high EC in the irrigation water, which lowers crop output. This implies that even when the soil is moist, there is less water accessible for plants when the EC in the water is higher; in other words, as EC rises, less water is available for irrigation by plants (Ogunfowokan and Obisanya, 2013). To emphasise, salinity and SAR regulate the infiltration rate; low salinity will decrease infiltration, while high salinity will increase it. Additionally, excessive levels of Na+ cations in the soil can be detrimental because they can reduce the permeability of the soil through adsorption of Na+ into the soil cation exchange and destroy the soil structure, making it compact and water-impervious (Hosseinifard and Aminiyan, 2015). As a result, SAR displays the impacts of soil Na+ concentrations as they build up. Researchers use the salinity diagram in Figure 2.5 to classify water to determine whether groundwater is suitable for irrigation. According to Suarez et al. (2006), the formula for computing SAR is as follows:

$$SAR = Na^{+} \div \frac{\sqrt{Ca^{2+} + Mg^{2+}}}{2}$$

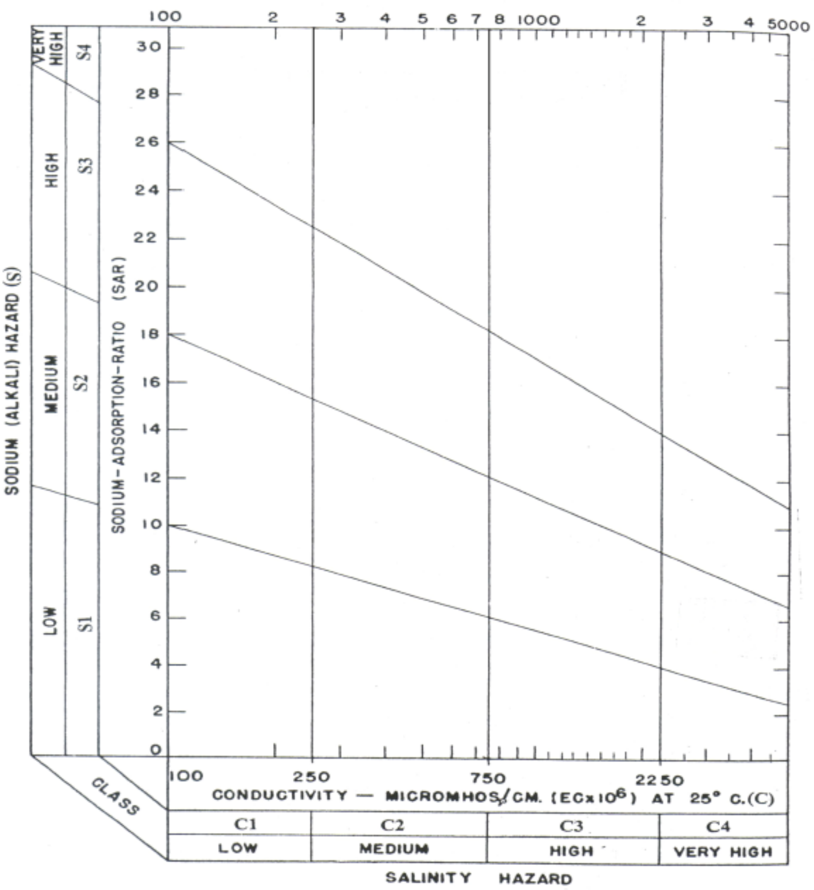


Figure 2.5 Salinity hazard diagram *(Source: El-Defan et al., 2016)*

Srinivasamoorthy et al. (2014) studied irrigation water quality in an Indian basin. Analytical data results plotted in two seasons (dry and wet) revealed that the water fell into the C3S1 zone that had high salinity and low Na content. They concluded that the water was good for irrigation with the least danger due to low exchangeable Na, hence, the Na hazard. Similar findings were obtained by Kenniche et al. (2022) when groundwater samples were located in the C3S1 zone. From the two studies it can be noted that the Na hazard is the most dangerous indicator to plant growth and productivity. In contrast, Hosseinifard and Aminiyan. (2015) plotted in the C4S3 area, which is representative of a field with significant Na hazard and very high salinity, and found the water to be unsuitable for irrigation.

2.6 Soil quality

Soil quality is one of the most significant environmental factors, after water and air quality, which have been researched for many years. The ability of the soil to function effectively in an ecosystem and the use of land, including the sustenance of biological productivity, environmental quality, and the promotion of plant and animal health, are also factors associated with soil quality, although researchers are primarily concerned with the potential or degree of pollution when studying the quality of water (Bünemann et al., 2018). Furthermore, soil quality, as defined by the National Research Council [NRC] (1993) as the capacity of soil to promote plant growth by regulating water infiltration, prevents water pollution by restricting potential pollutants such as organic wastes and agricultural and industrial effluents. Since soil is used for many land use activities, it reacts very slowly to land use changes, hence making it more difficult to accurately detect these changes, unlike water quality changes which happen abruptly.

Determining the characteristics of the soil that allow it to be sustainable to support land use and economic development is part of the evaluation of soil quality. Similar to the quality of water, it is affected by both natural (geological) and man-made activities that significantly change its chemical and geochemical properties, rendering it unfit for usage. The functioning of landfills is one example of a human activity that has been shown to affect the quality of the soil where it is located (Augustin and Viero, 2012). To put it into perspective, the distribution of particle and grain sizes in the soil structure is a key factor in determining whether or not the soil will be contaminated by landfill effluents. Additionally, the chemical and geochemical nature of the soil must be identified to quantify the level of pollution; for this reason, the following section goes into great detail about these three factors.

2.6.1 Particle size distribution

Humus, weathering, and leaching are some of the natural processes that affect soil formation; weathering is the most significant (Szecsödi et al., 2021). Therefore, the mixture of primary and secondary minerals that results from weathering and fragmentation can be referred to as soil. The type of minerals that are integrated into the soil medium determines the size of the mineral particles. According to Nyika et al. (2019), the distribution of particle and grain sizes is expressed as a percentage of the total weight of the dry soil assigned to a particular particle size. The type of soil-forming processes, for example, sedimentary deposition, has a direct impact on the particles that are generated, from where, as a result, Particle Size Distribution (PSD) is derived. PSD influences the pore spaces and the activities that take place along them. It also shows the texture of the soil and the relative positions of the particles. Yong et al.

(2017) mentioned that PSD also controls the dynamic properties in surface processes such as debris flows, landslides, and avalanches. In addition, PSD is very useful to comprehend the genesis of the soil that involves the erosion, transport, and deposition of sediments, determination of trends that relate to surface processes, stability of the slope, and reaction of particles with liquids. Additionally, PSD provides information about the cation exchange capacity of the soil, which regulates sorption and mineral assemblage (Ngole Jeme and Ekosse, 2015).

At landfill sites, several researchers have studied the PSD of polluted soils. In their 2019 study, Nyika et al. examined a landfill in the South African province of the Eastern Cape and found that, in contrast to coarse-grained soil, the soil medium was primarily composed of fine clay that is interlayered with silt. The presence of Pb, Cu, Cr, and Cu metals supported the idea that fine clay, which has a higher surface area and is indicative of high adsorption, might retain heavy metals and minerals in soils. Another study by Ololade et al. (2019) found that soil profiles with particle PSDs typical of clay to sandy clay and clay loam to loamy sand were linked to increases in heavy metal contents. Once more, the existence of clay forms in the soil layers is the reason for this. Metal concentrations were lower in the loamy to sand soil layers. Akortia et al. (2019) discovered that soil samples with greater grain sizes had higher quantities of TOC (150 μm – 250 μm (micrometres) when compared to smaller ones (45 μm – 150 μm) in the grain size range. This supports the accumulation of heavy metals in smaller clay particles instead of larger PSD.

2.6.2 Chemical and geochemical characterisation

The chemical and geochemical properties of the soil and soil stratigraphy must be thoroughly studied to assess the quality of the soil for a variety of land-use applications, including industrial, agricultural, and commercial use. Three fundamental processes determine the chemical makeup of soils: primary mineral dissolution, secondary mineral precipitation, and solution dilution (Abdennour et al., 2020). This is due to the introduction of water (aqueous liquid) into the soil horizon, which induces chemical reactions with mineral phases present in the soil to bring about this change. In South Africa, which has a wide extension of geological nature classified by a variety or rocks and soils, the geochemical composition of soils is largely impacted by natural processes and anthropogenic activities (Oyebanjo et al., 2020).

It has been determined that due to a change in the geochemical nature of soil, the agricultural potential has been affected; for instance, the nutrient availability of the soil was deficient for plant growth. Because of the high or low concentrations of necessary components in the soil, the food that is vital to human survival was unfit for human consumption (Oyebanjo et al., 2020). As the intensity of elements leaching out of Brazilian soils rose, Silva et al. (2016) also reported that weathering was the source of the geochemical association of major and trace elements. At landfill sites, which are human-activity-driven, chemical and geochemical changes have also been reported. It should be noted that soil geochemistry is mostly associated with pH, heavy metals, TOC and exchangeable ions, which were the focus of this study. Kuraieva et al. (2021) analysed the geochemistry of the polluted soils by the landfill waste. When compared to soils from the area that were not affected by the leachate contamination, they discovered that the soils contained high quantities of heavy metals. The low levels of exchangeable cations (Ca_2+, Mg_2+, K+, and Na+), organic carbon content (%Corg), and pH of contaminated soil compared to non-polluted soils, which had high concentrations, all supported the idea that the sorption capacity of anthropogenically polluted soils is lower than that of natural soils. Similar results were published by Azimov et al. (2020), who found that the chemical and geochemical properties of contaminated soils indicated elevated levels of heavy metals (Cu, Zn, and Pb) in the area surrounding the landfill site. However, the concentrations of exchangeable cation content, pH and carbon in the soils that were collected close to the landfill site were lower than the ones collected away from the landfill site, owing to the decrease in sorption capacity of loam soils as opposed to clay soils. Moreover, Ololade et al. (2019) made a different observation on clay soils, which had an increase in heavy metal concentrations together with exchangeable cations, pH and organic carbon content due to a high adsorption capacity of clay soils as opposed to a low adsorptive capacity of loamy soils in the landfill site.

The degree of contamination or pollution of the soil is hinged by the type of soil which possesses various physical (grain size, structure, mineralogy) and chemical or geochemical properties that can either favour or disfavour pollution from anthropogenic and natural processes. Certain soils can act as buffers (e.g. clay), thus preventing pollutants from passing through them by holding them in place, while others can easily allow the passage of pollutants into either the groundwater source or the plant material it is in contact with. It is with this urgency that the soil quality be clearly determined and understood in different land use in order to decipher its suitability of use.

2.6.3 Soil pollution indices concepts

Elevated concentrations of heavy metals in soils are strongly related to their geoaccumulation, bioaccumulation, and transit rates, which reveal information about their origin (Kowalska et al., 2018). Heavy metals can originate from both domestic and foreign sources, and they can accumulate in agricultural soils in large quantities, creating environmental problems that affect food supply and security. Because of this, pollution indices have been utilised extensively as a useful instrument for evaluating the condition of the soil (Ahirvar et al., 2023). Utilising pollution indices aids in assessing the degree of soil deterioration and environmental risk. They are also useful in determining the natural or human origin of heavy metals, and they are essential for monitoring soil quality to guarantee its sustainability for agricultural use. The Geochemical Background (GB), which was created to distinguish between abnormal and natural concentrations of heavy metals in the soil environment, is a component of the computation of soil pollution indices (Kowalska et al., 2018). Several authors have given definitions of the GB. For example, Nganje (2010) and Karim et al. (2015) described it as a metric that distinguishes between the concentrations in soil that are naturally occurring and those that are caused by human activity. It is the “normal abundance of an element in barren earth material,” according to Hawkes and Webb (1962). Furthermore, according to Matshullat et al. (2000), it is a “relative measure to distinguish between the concentrations of natural elements and the anthropogenically influenced concentrations in real sample collections”. Individual indices (contamination factor, ecological risk factor, enrichment factor and geoaccumulation index), and complex indices (Nemerow pollution index, degree of contamination, ecological risk index, and average pollution index) are the two primary categories into which pollution indices are divided. While individual indices are computed for each heavy metal, complex indices require a thorough, integrated assessment that encompasses multiple heavy metals.

2.6.4 Commonly used soil pollution indices

- **Contamination factor**

By identifying the heavy metal concentration of the soil surface and the pre-industrial values given, the Contamination Factor (CF) enables accurate evaluation of soil contamination (Kowalska et al., 2018). Advantages of the CF are that it is a direct and simple method that provides the difference between the sample as well as background values. It gives a precise scale in its determination. However, its downsides are that it does not need natural processes to vary, no GB values are included, and it excludes the ability of heavy metals (Inengite et al., 2015). It is calculated by the formula:

$$C_f = C_m \div C_{p-i}$$

Equation 2.9

Where Cp-i is the industrial reference value and Cm is the mean heavy metal concentration obtained from a minimum of five samples. Based on the level of contamination, the contamination factor is categorised as follows:

- C<1 indicates a low level of contamination.
- 1≤ C< 3 indicates a moderate level of pollution.
- A significant contamination factor is 3≤ C<6,
- C>6 indicates a high level of pollution.

- **Geoaccumulation index**

According to the precise GB, the geoaccumulation index permits heavy metal contamination in either the A or O horizons (Muller, 1969). Being a popular index, it helps compare contamination from the past and present in an easy-to-understand manner. According to Kowalska et al. (2018), it also includes precision in size and its 1,5 constant limits variations in lithology. The following formula is used to calculate the geoaccumulation index: Base 2 logarithm of background concentration and heavy metal concentration:

$$\text{Geoaccumulation index} = \log_2[\frac{\text{Cn}}{1{,}5\text{Bn}}]$$

Equation 2.10

Where Cn represents the measured heavy metal concentration, Bn represents the metal geochemical environment concentration, and a value of 1,5 was chosen due to the lithological variability. It consists of a range of pollution classes where the geoaccumulation index of class 0 represents unpolluted and a class greater than 5 is extremely polluted (Inengite et al., 2015).

- **Enrichment factor**

As a reference in GB and examined samples, it is a measurement of the impact of human activity on heavy metal concentrations in the soil environment that is indicative of low variability of occurrence (Kowalska et al., 2018). The Enrichment Factor (EF) is used for its strong estimation capacity in anthropogenic impact and assessment of the origin of heavy elements. It also has a scale that is precise and allows for reduction in metal variability (Sayadi et al., 2015). Its weaknesses are that it often assess above the uncontaminated concentrations and is measured in relation to reference values. Al, Fe, Ca, Ti, Sc, or Mn are reference elements used in the formula:

$$EF = \left[\frac{Cn}{LV}\right] sample \div [\frac{GB}{LV}] background$$

Equation 2.11

Cn represents heavy element concentration while LV is one of the elements Mn, Fe, Ca, Ti, and Sc. Al is represented by the V backdrop. EF values between 0, 5 and 1, 5 suggests natural cause for heavy element enrichment in soil, whist the EF exceeding 1, 5 suggests an anthropogenic source.

- **Single Pollution Index (PI)**

It can be utilised in intricate index computations and identifies the heavy element that most threatens the soil ecosystem with contamination (Kowalska et al., 2018). It is expressed using the formula:

$$PI = \frac{Cn}{GB}$$

Equation 2.12

Where GB is the geochemical background value and Cn is the amount of heavy metals in the soil medium.

2.6.5 Sum of contamination (PI_{sum})

It is a commonly used complicated index to measure the amount of heavy metals in soil. According to Kowalska et al. (2018), PI_{sum} is the total of all measured concentrations of heavy metals in the soil. It also has benefits, such as the capacity to compare contamination in different soil ecosystems and incorporate all heavy metals that have been analysed (Inengite et al., 2015). Despite its advantages, it also has disadvantages, namely that it does not need differences in natural processes and leaves out heavy metal availability. The formula for calculating the PI_{sum} is given as:

$$PI_{sum} = \sum_{i-1}^{n} PI \qquad \textit{Equation 2.13}$$

Where n is the number of heavy metals and PI is the single pollution index's computed value.

- **Nemerow Pollution Index ($PI_{NEMEROW}$)**

The popular NPI technique has been used to evaluate the total amount of heavy metal contamination in soil, including in the A and O horizons. It provides a range of advantages by directly highlighting specific contaminated elements in soil, thus reflecting the soil pollution (Oti, 2015). Further, it is widely used as it gives an accurate scale based on PI values and considers all individual elements. However, it has limited disadvantages which entails the absence of a weighing factor and element ranking based on contamination. It is computed using the formula below.

$$NIPI = \sqrt{\mathrm{Plave}^2 + \mathrm{Plmax}^2} \qquad \textit{Equation 2.14}$$

Where $\frac{\mathrm{Cheavymetal}}{\mathrm{Cbackground}}$, Plave^2, and Plmax^2 represents the mean as well as the highest PI of every metal. The NIPI is classified from no pollution to extremely severe pollution.

- **Pollution Load Index**

Pollution load index (PLI) is a quick and easy to use method that shows how the soil is deteriorating at which extend as a result of heavy metal accumulation. Similar to Nemerow's complex method, it also makes a combination of a certain number of analysed metals and promotes the comparison of contamination at various soil areas (Sayadi et al., 2015). Additionally, it can identify how frequently heavy metal concentrations in soil exceed their natural levels. The downside of this index is that it does not need a variability of natural processes with respect to GB and it excludes heavy metal availability. It is computed by the formula:

$$PLI = \sqrt[n]{PL1 * PLI2 * PLI3 * \ldots\ldots PLIn} \qquad \textit{Equation 2.15}$$

Where PLI stands for the computed values of the single Pollution Index and n is the number of heavy metals that were analysed.

2.7 Overview of groundwater vulnerability

Vulnerability studies have the objective of differentiating clearly where groundwater is more vulnerable to contamination and in areas where it is least vulnerable, given the land use activities that are taking place in an area. According to Oke and Fourie (2017), the origin-pathway-receptor/target model is essential for determining the risk of pollution from contaminants when evaluating groundwater vulnerability. The pathway is the medium through which the contaminant travels, which is essentially the earth's crustal material or lithology. The target is the final destination, the water table, the groundwater body that underlies the pathway in the saturated zone. The origin describes the source of the contaminant from where it is released, for example, from the surface as a result of precipitation (Oke and Fourie, 2017). Vulnerability can then be defined as the susceptibility of the aquifer to pollution, which can reduce the quality of groundwater (Hosseini and Saremi, 2018). The National Research Council [NRC] (1993) defined vulnerability as the likelihood or the tendency of the pollutant to reach a certain location in groundwater after being introduced from some location in the upper surface. Furthermore, groundwater risk, which poses a hazard to human health and the environment since it already involves the existence of the contaminant, is not the same as vulnerability, which calculates the probability that the pollution will occur.

Vulnerability can be classified into three different types, which are unique in terms of their approach in its determination. The first type of vulnerability is known as intrinsic vulnerability, and it refers to the innate vulnerability to contamination that is supported by physical environmental elements independent of the characteristics of the pollutant (Hosseini and Saremi, 2018). The second is the specific vulnerability, which focuses on the contaminant in terms of its transport properties through the subsurface, and the third group is specific to the simplification of complex geohydrological information for decision-makers and the public. Additionally, vulnerability methods can be broadly categorised as overlay, process-based, statistical, index-based, and process-based (Santhosh and Babu, 2018). Since both index and overlay approaches are non-measurable and require allocating weights to significant elements, they are frequently employed, particularly in sub-Saharan African nations with limited data. Technologies such as GIS are then utilised to create vulnerability maps. Numerous scientists have developed a variety of vulnerability approaches, such as the RTt, GOD, and DRASTIC methodologies, to assess an area's susceptibility to pollution. The next sections provide a detailed discussion of the aforementioned techniques.

2.7.1 DRASTIC method

USEPA created the DRASTIC model as an approach to assess groundwater vulnerability in any geohydrologic scenario (Oke, 2020). The seven criteria that comprise its name, DRASTIC, are the basis for its evaluation and are as follows:

- **D**epth to the water table, where the likelihood of pollution increases with increased depth.
- Net **r**echarge, which is determined by the volume of water that enters the water table through the lithology. As it infiltrates, the net recharge may potentially transport pollutants in solution.
- **A**quifer media, which is in charge of lowering the pollutants as they permeate the ground. It is made up of the aquifer's physical characteristics.
- **S**oil media stands for the soil media that controls the quantity of water that seeps into the ground and is situated on the upper horizon in the vadose zone.
- **T**opography, or the surface's slope or gradient. Steeper slopes will promote the runoff of water more than gentle slopes.
- Like the aquifer media, the **i**mpact of the vadose zone regulates the flow and minimises the amount of pollutants that can enter the saturated zone and contaminate the groundwater. Its physical characteristics are crucial in controlling the amount of water that seeps in.

- The term conductivity or hydraulic conductivity describes how easily water may move through the aquifer media. It establishes the rate of the water.

These physical characteristics are what regulate the aquifer’s propensity for pollution. According to Oke (2020) and Santhosh and Babu (2018), the drastic model operates under a series of presumptions that distinguish it from all other vulnerability models. These presumptions are as follows: The pollution must originate at the surface, enter the saturated zone by precipitation, move at the same speed as water, and be unconfined in shallow aquifers. Santhosh and Babu (2018) also mentioned that the ignorance of the method to the fractured aquifer is a disadvantage, and that the area being assessed should be 100 acres or larger. Though it can have disadvantages, there is an advantage to the method, which is its simplicity and straightforwardness. The data needed for use is mostly accessible and the results can be easily interpreted in comparison to other methods, thus making it more widely used. Now, in order to calculate the vulnerability of the aquifer to pollution, the drastic index is used by apportioning the weights and ratings of all the seven parameters based on their importance in attenuating the contaminant that is infiltrating. The weights are in the range of 1 (least significant) to 5 (most significant), while the ratings are ordered from 1 (least important) to 10 (most important) (Nnadozie et al., 2019). After the allocation of weights and ratings follows the sum of all the products of all the parameters that make up the drastic method, and it is as follows:

$$\text{Drastic index} = D_R D_W + R_R R_W + A_R A_W + S_R S_W + T_R T_W + I_R I_W + C_R C_w \qquad \textit{Equation 2.16}$$

Where, DRASTIC represents all seven parameters of the model, *r* is for rating and *w* is for the weight.

The determination of vulnerability; however, does not end with just identifying the drastic index, therefore, studies have identified a measurement tool in the form of ranges to determine whether a certain aquifer falls into high vulnerability or low vulnerability areas. Santhosh and Babu (2018) delineated the ranges as follows: very less vulnerable (90−110), less vulnerable (110−120), moderately vulnerable (120−130), highly vulnerable (130−140), and very highly vulnerable (140−162). Also, in calculating the vulnerability of groundwater from the DRASTIC parameters it can be determined which parameters pose the highest risk of pollution and which ones play an integral part in attenuating it. Parameters with a high risk and attenuation factor will be allocated higher ratings than others.

2.7.2 GOD method

GOD method is an alternative strategy that uses an index methodology similar to DRASTIC, but it solely takes into account the groundwater occurrence , the aquifer's overall lithology, and the depth to the aquifer/groundwater (Nnadozie et al., 2019). When compared to the DRASTIC model it does not possess many parameters, therefore, even scientists place it as a second choice in terms of result delivery and data accuracy, and that is why they use this model in conjunction with others. Deubalbe et al. (2021) added that the GOD method measures the vertical infiltration of the pollutants without the consideration of the lateral movement into the saturation zone. Also, the role of the soil in attenuating pollution in this method is neglected, and it allows for the rapid estimation of vulnerability in comparison with other methods. Using this procedure, the ratings are assigned to each of the parameters, and the GOD index is then calculated by multiplying the ratings, using the following formula.

$$GOD\ Vulnerability\ Index = G \times O \times D$$ *Equation 2.17*

This method uses a variety of vulnerability classes to assess the area's susceptibility to pollution. These classes are very low vulnerability (0−0,1), low vulnerability (0,1−0,3), moderate vulnerability (0,3−0,5), high vulnerability (0,5−0,07), and extreme vulnerability (0,7−1,0). Salim et al. (2019) have utilised the GOD approach to evaluate the level of vulnerability surrounding land-use activities such as farms, industries, and disposal sites.

2.7.3 Rainfall travel time method

The RTt method, just like the GOD method, does not contain a larger number of parameters like the DRASTIC method and several others. Because of its simplicity, it can be easily used in places with limited data, particularly in developing countries and sub-Saharan Africa. In its principal description, it is a method that is based on the recharge of groundwater via rainfall, with the assumption that rainfall is the factor, and that there is a vertical infiltration of the water through lithological layers (Oke et al., 2016). As implied by its name, the RTt necessitates determining the area's rainfall, ensuring that the seeping contaminants move at the same pace as the water, and taking into account the characteristics of the lithology that the water percolates through (Oke et al., 2016). The RTt model also has some disadvantages or limitations which are mainly the following: It does not consider the anthropogenic activities and it does not cater for assessment in different geological settings such as the DRASTIC model. The environments include karst areas dominated by carbonate rocks that display different flow characters like turbulent flows. Also, the large surface waters are not allocated in the method, together with the destination of pollutants as they seep into the ground. The method does not also consider the underground pollution sources such as sewage leakages, petroleum stations that store fuel, as well as other underground pollution sources.

The RTt method, like DRASTIC and GOD, has a similar way of determination by the use of weights and rating assignment of each of the parameters based on their importance. But before its formula can be computed it is important to outline that its parameters were chosen for a specific reason, just as with other models, in the following way. According to Daly et al. (2002), the R parameter was chosen because precipitation is cheap, simple to collect from various locations, and readily moves from the surface to groundwater. On the other hand, the Tt parameter measures how quickly a liquid moves through the surface of the earth and into the water table, which can be related to hydraulic conductivity, in which an increase in conductivity means that the water flows at a higher rate through the rock strata such as sandstone and gravel (Santhosh and Babu, 2018).

To calculate the RTt of a contaminant as it flows through the soil horizon, the rainfall of an area per year has to be determined, as well as its travel time through the rock layers. Regarding the travel time, the formula according to Saayman et al. (2007) is as follows:

$$T_{time} = Z \times \theta \div V_d$$ *Equation 2.18*

Where T_{time} is the number of years that the trip takes; V_d is the average recharge in metre per day; Z is the thickness of the vadose zone in metre (m); and θ is the moisture content of the vadose zone. In both worn and sedimentary rocks, the migration of pollutants into unconfined aquifers can be tracked using transit time (Oke and Fourie, 2017). The RTt can be calculated as follows once the rainfall and journey time have been determined:

$$RTt = R_R R_w + Tt_R Tt_w$$ *Equation 2.19*

2.7.4 Aquifer vulnerability index method

The Aquifer Vulnerability Index (AVI) method was developed in Canada to give an estimation of aquifer vulnerability by considering the thickness of sedimentary layers above the saturation zone and hydraulic conductivity (Moges and Dinka, 2022). This model has advantages that include no rating and weight use, few data needed in its calculation, and its suitability to be applied in land use management. However, since it utilises a limited amount of data it tends to ignore processes taking place in both soil and rock, and it excludes water quality. It is calculated using the following equation:

$$c = \sum_{i=1}^{n} . d_i \div K_i$$ *Equation 2.20*

Where d_i and K_i stand for the aquifer's thickness and hydraulic conductivity, respectively. The hydraulic resistance (c) represents the time it takes for a pollutant to move downward through a porous material in a process known as advection, according to Kumar et al. (2015).

2.7.5 Concentration of flow, overlaying layers, and precipitation method

This method was first discovered in Spain to map the groundwater vulnerability of Karst aquifers where the pre-existing rocks often dissolve over a period of time to form fissures, fractures, sink holes, and cracks in the surface and subsurface lithology (Goyal et al., 2021). Three parameters – concentration of flow (C), overlaying strata (O), and precipitation regime (P) – are used in the method's computation. For more emphasis, the C factor stands for the concentrated percolation caused by precipitation and surface runoff in karst areas. Conversely, the O factor represents the varied precipitation of different amounts and the groundwater protection provided by the strata above the water table (Goyal et al., 2021). As a widely applied method, its advantage is that it uses very simple parameters that are straight forward, but requires GIS software to process and plot its maps. It is calculated using the formula:

$$COP = C \times O \times P$$ *Equation 2.21*

2.7.6 SINTACS method

Groundwater vulnerability for climatic, socio-economic, and geohydrological variables in the Mediterranean region was evaluated using the SINTACS method (Rahimi et al., 2022). It is quite similar to the DRASTIC model since it also includes seven parameters: S – depth to water table, N – unsaturated, T – soil type, A – aquifer geohydrological features, C – aquifer hydraulic conductivity, and S – roughness of the land surface. The SINTACS abbreviation, however, is composed of italian words, and is technically a DRASTIC version used in the Intalian context and conditions. According to Jaunat et al. (2019), this method consists of a few advantages that such as its suitbility for use for wide land use activities like coal and oil-dominated area, and its cost-efficiency and simplicity. Similar to the DRASTIC model, it entails the element of subjectivity in weightings and ratings of parameters. Is is calculated in the following way:

$$SI_v = \sum_{i=1}^{7} . \sum_{j=1}^{n} (P_i W_j) \qquad \text{Equation 2.22}$$

Where W_j is the weight of the jth weight classification, and P_i is the ith parameter. The SINTACS approach states that the susceptibility increases with the SI_V.

2.7.7 EPIK method

One of the earliest models to evaluate groundwater vulnerability in karst situations was the EPIK model. The EPIK model is composed of four parameters: Epikarst (E), the protective cover (P), the infiltration conditions (I), and the Karst network (K). This method is suitably designed and applied in carbonate rocks such as dolomite and limestone, as well as soluble rocks that are typical of rapid turbulence (Moges and Dinka, 2022). It is beneficial to use since it encapsulates more parameters and low ratings; however, its downsides are that it is limited to karstic conditions, and needs more detailed geophysical investigations that are expensive and time-consuming. The EPIK index is calculated using the following formula:

$$EPIK_{Index} = (\alpha \times E) + (\beta \times P) + (\gamma \times I) + (\delta \times K) \qquad \text{Equation 2.23}$$

It has recommended weights that are given as α=3, β=1, γ= 3, and δ=2, with a vulnerability index range of between 9 and 34, respectively.

2.7.8 PI method

The PI method is another karstic method applied in karst aquifers for vulnerability assessment. It was created by Goldscheider et al. (2000) and utilises the transfer of contaminants into groundwater from the surface sources via the unsaturated zone. It highlights the inherent susceptibility of the karstic aquifers and takes into account two parameters: the Protective cover (P) and the Infiltration (I) (Moges and Dinka, 2022). The P factor explains the protection factor that stems from the thickness of the strata, while the I factor is the infiltration properties of the beds due to fluid flow (Oke, 2016). The PI method is beneficial in that it accommodates various hydrogeological settings and can determine the intrinsic vulnerability of karstic aquifer systems. However, it does not give attention to attenuation conditions. It is calculated as follows:

$$P = \left(B + \sum_{i=1}^{m} . M_i G_i\right) + \sum_{j=1}^{n} . B_j M_j)\, W + Q + HP \qquad \textit{Equation 2.24}$$

In this case, B stands for effective field capacity, M_i and M_j for soil and bedrock stratum thickness, G_i for effective subsurface stratum protection, W for recharge infiltration rate, and HP for hydraulic pressure conditions. The groups into which the PI technique is divided are P=1, which denotes a low degree of protection, and P=5, which denotes thick and protective layers.

2.7.9 SI method

The Susceptibility Index (SI) method was developed to assess groundwater vulnerability for both medium and large regions (for example, between 1:50 000 and 1:200 000). Hilal et al. (2024) evaluated many potential hazards to groundwater, such as fertilisers, land use, and anthropogenic activities. The parametric technique, comparable to the DRASTIC model, assesses the vertical susceptibility of groundwater contamination from agricultural activities. Based on the DRASTIC model, this model considers four parameters: water table depth, recharge, aquifer media, and terrain, as well as Rebeiro’s (2000) land use parameter. It evaluates a variety of factors, including fertilisers, land use, and human activities as potential hazards to groundwater (Hilal et al., 2024). It is a parametric approach comparable to the DRASTIC model that assesses the vertical susceptibility of groundwater contamination, primarily from agricultural operations. Based on the DRASTIC model, it uses four parameters: depth to the water table, recharge, aquifer media, and terrain, as well as an extra parameter of land use established by Rebeiro (2000). The method is suitable for agricultural areas that are composed of vertical percolation of NO_3^- and pesticide pollutants but is subjective in both weighting and rating (Moges and Dinka, 2022). The SI index is determined using the following formula:

$$SI = D_C \times D_P \times + R_C \times R_P + A_C \times A_P + T_C \times T_P + O_{SC} \times O_S \qquad \textit{Equation 2.25}$$

Where D is the depth to the water table, R is the recharge, A is the aquifer medium, T is the terrain, and O is the land use. Additionally, the ratings of the first four parameters are given by multiplying them by 10, and the weighted and rates parameters give the overall vulnerability by adding them together.

All three vulnerability methods have been applied extensively, especially the DRASTIC and GOD methods, although DRASTIC is the most widely used of all. Researchers have shown that the methods are advantageous and disadvantageous in their different ways, but when they are mostly reliable in their results, especially when they are used in conjunction with one another. Also, vulnerability methods are able to give the estimates of the potential groundwater pollution that can take place in an area and are useful for decision-makers, governments, the public, as well as scientists. It is needful to say that their graphic depiction by means of GIS technology upon calculation makes it easier to read and interpret, and though there are many of these vulnerability methods being formulated for various purposes, this study has specifically chosen the above-mentioned three methods.

2.7.10 GALDIT method

Chachadi and Lobo-Ferreira (2001) created the GALDIT method to address groundwater vulnerability in coastal areas prone to heavy seawater intrusion. The approach is made up of six parameters, three of which emphasise the characteristics of coastal aquifers, namely, G – aquifer type, A − hydraulic conductivity, L − height above sea level, and D − distance to sea coastline, I − qualitative sea water advancement impact on coastal areas, and T − thickness of the aquifer (Rahimi et al., 2022). The GALDIT method has been discovered to be advantageous due to data acquisition simplicity and availability, but one of its fallouts is that it does not pay attention to the impact of groundwater pumping on seawater intrusion, which can result in salt water to be drawn close to the well (Goyal et al., 2021). It is calculated in a simple formula as follows:

$$SI = G_R G_W + A_R A_W + L_R L_W + D_R D_W + T_R T_W + I_R I_W \qquad \textit{Equation 2.26}$$

The method entails the rating and weighting of parameters like any other methods to compute the overall index.

2.8 Research gap

This study has identified two major research/study gaps from the prior research and existing literature that it sought to address as its main purpose and or objective.

The first gap identified based on past research was the population gap. The municipal landfill in the Botshabelo Township, Free State province, has not been investigated for the influence of leachate on water (surface and groundwater) and soil quality. The landfill site had been in operation for several years and has seen an increase in the amount of waste placed on it, as a result of the township's expanding population. There are important aspects surrounding the landfill site that could be the drivers of pollution, such as the type of landfill, leachate collection and treatment systems, and the liner and waste cover systems. The majority of the research has only been undertaken in metropolitan regions, including Bloemfontein, which was conducted by Ololade et al. (2019) and Rinae et al., (2020), leaving the less developed townships unstudied. Similar research has been undertaken in Gauteng, the Western Cape (Cape Town), and KwaZulu-Natal provinces (Elumalai et al., 2017; Osibote and Rabiu, 2016; Sibaya et al., 2017). The investigation on this landfill site in Botshabelo was important because of the growing population and the economic contribution in the Mangaung district.

Second, based on the literature studied, a methodological gap also existed. The use of vulnerability methods in assessing the groundwater vulnerability at landfill sites has been lacking, particularly in South Africa. For the vast majority of studies, Babika et al. (2022); Oke and Fourie (2017); Oke et al. (2016) and Rukmana et al. (2019), vulnerability methods have been widely applied in other areas than in the solid waste management landfill sites. Also, even though they have been applied at landfill sites by Morita et al. (2021) and Santhosh and Babu (2018), the methods have not been used in combination with one another, for example, only DRASTIC and GOD, or RTt have been used on its own. The importance of using different methods in assessing groundwater vulnerability to pollution helps to generate more accurate and reliable data through a wide range of interpretation. Furthermore, a study was conducted by Van Niekerk (1996) in the same area as this study, but it did not cover the aspects of pollution indices, surface water and soil quality, and groundwater vulnerability. This study, then, sought to close this gap by addressing these aspects in a detailed manner, and as a result, sought to holistically contribute more scientific knowledge and bring understanding for better decision-making by stakeholders.

Chapter 3
Materials and Methods

3.1 Introduction

This chapter provides a summary of the research study area in terms of the regional setting, geology and geohydrology, type of soil, climate, and terrain. This chapter further delves into much greater detail pertaining to the waste management practices that are operated, as well as a background study of the locality. The methodology and study design are discussed in detail in the latter part of the chapter.

3.2 Study area

The Botshabelo solid waste landfill site was chosen for this study. The MMM manages the site, which is located in the Botshabelo Township, Free State province. The landfill site is located close to the Pirana Mountains as it lies on the flat land surface. Also, it is located close to the abattoir, and the surrounding land is purposed for agricultural activities.

3.2.1 Regional setting

The Free State province is located in the flat area of South Africa's heart or centre (Free State Provincial Government, 2013). It is surrounded by provinces other than the Western Cape and Limpopo (Figure 3.1). On the eastern side, it shares an international border with Lesotho, forming a bean-like shape. It is part of the Great Escarpment, which separates it from KwaZulu-Natal and the Eastern Cape. The province encompasses 10,6% of the total surface area of the country, or around 129 464 km^2, and is the third largest of the nine provinces. The province is divided into five districts, one of which is the MMM, wherein the study area is located. The Motheo district is part of this municipality. Bloemfontein, Botshabelo, and Thaba'Nchu merged in 2000 to become part of the municipality's integrated waste management plan (Mangaung Metropolitan Municipality, 2011). Bloemfontein serves as the economic hub for the province, while Botshabelo is regarded as the province's and one of South Africa's fastest-growing towns. The population increased by approximately 1,48% between 2001 and 2011, from 645 440 to 747 431 (Statistics South Africa, 2011).

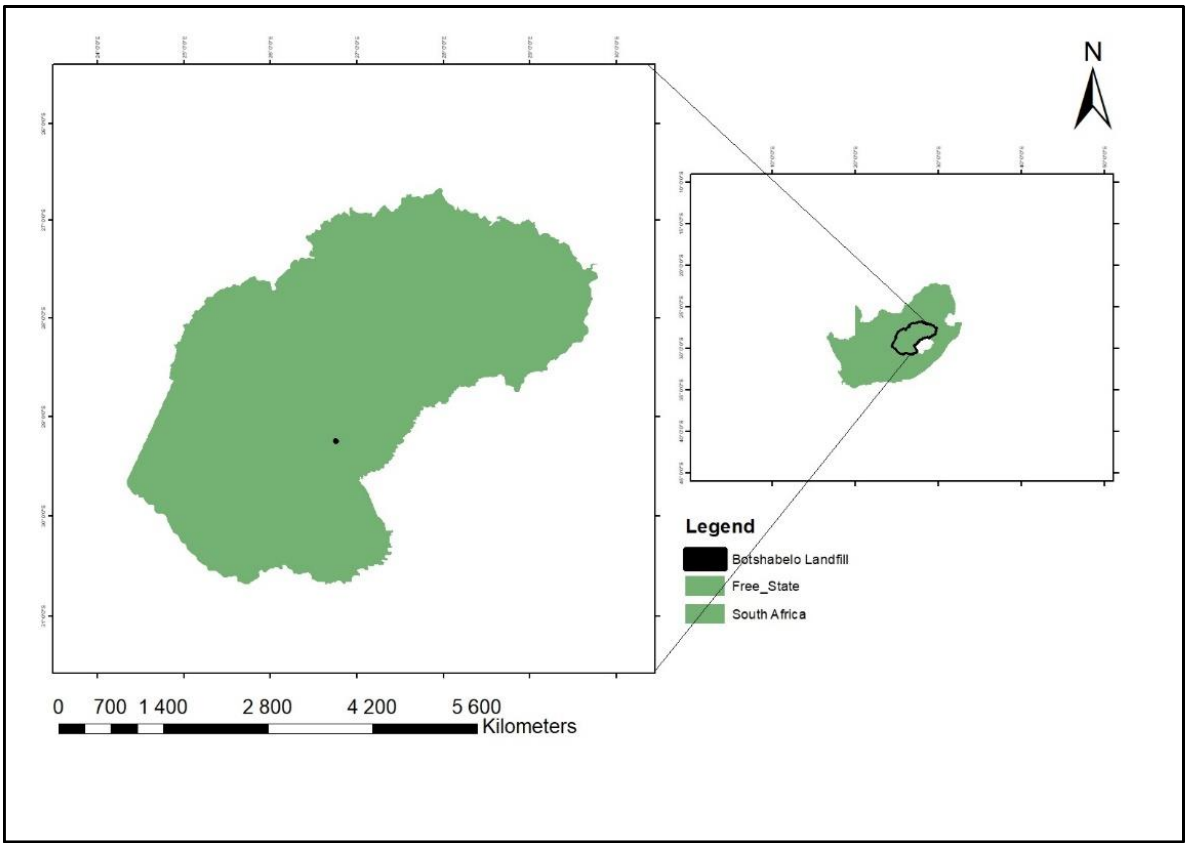


Figure 3.1 Regional map of the Free State *(Designed by author using ArcGIS, version 10)*

3.2.2 Climate

Botshabelo is located in South Africa's summer rainfall area, which is classified as a sub-humid, warm zone with an annual water deficit (Mangaung Metropolitan Municipality, 2020a). During summer, the area is characterised by thunderstorms as well as soft rains of approximately equal amounts. Botshabelo has a mean yearly rainfall amount of 533 mm, and its average temperatures can range between 30 °C at maximum in January, and the minimum temperatures approximate 1 °C in June (Figure 3.2). The monthly evaporation rates of the area are elevated in the summer season, approximating 323 mm in December, whereas in winter, they are about 85 mm in July.

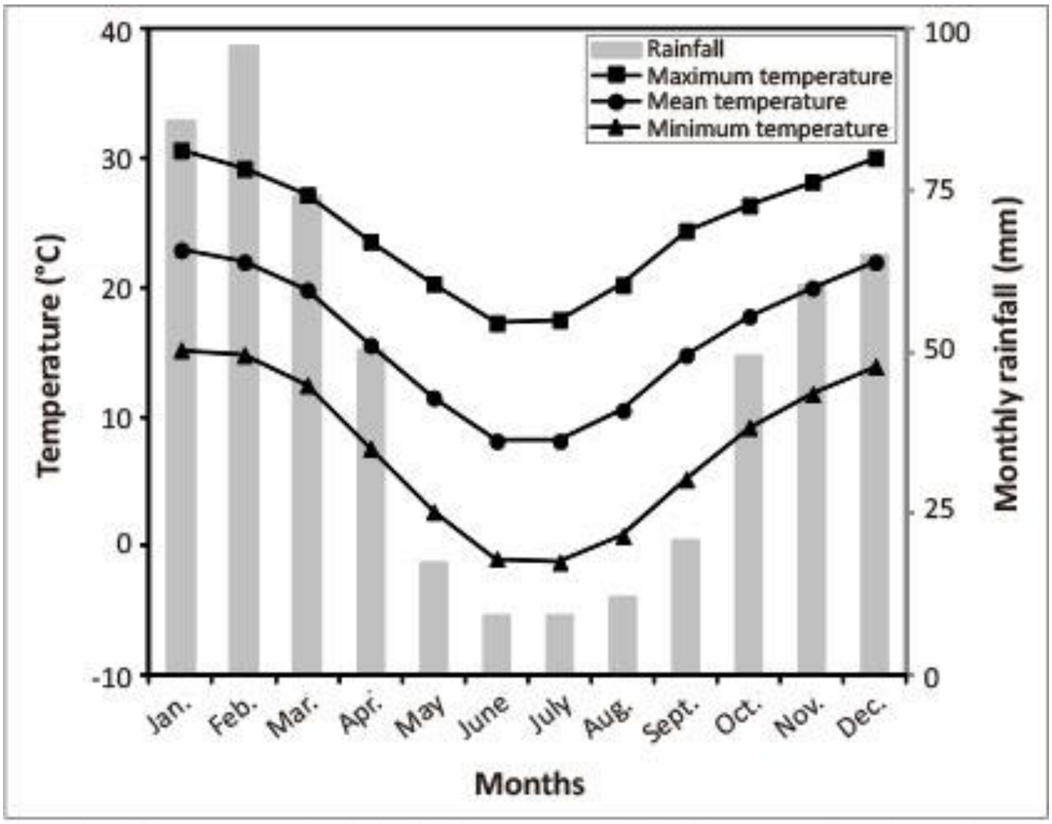


Figure 3.2 Climate map of the Free State *(Source: Dingaan and Du Preez, 2013)*

3.2.3 Topography

Botshabelo is 55 km east of Bloemfontein, and it covers approximately 6 180 ha in extent (Mangaung Metropolitan Municipality, 2020b). The occurrence of big hills, mountains, and ridges is noticeable as it is located in the eastern section of the Free State province. The Klein-Modder River is the most visible natural feature that runs through the Botshabelo township, and from the topographical view and aerial imagery, the township is characteristic of the slope that is flat, undulating, and towards the river it slopes. The township has ridges that run along the eastern side in the form of a range and along the western side. The highlands or ridges are sandstone terrains. The height above sea level, altitude of the area differs from 1 680 m on the eastern side, particularly along the system of ridges, and decreases immensely to 1 350 m along the western side. This creates a difference in the elevation to signify that the topography is indicative of mountainous areas.

3.2.4 Soils

It is a well-known concept that the geology of a place and the natural processes of erosion and weathering determine the kind of soil that is generated because soils are the product of an area's rock type. The type of soil that are derived from the area include blackish or dark-brown silty-clay, with some lime and or ferricrete nodules (Mangaung Metropolitan Municipality, 2020b). The Botshabelo type of soil is, therefore, typical of a high content of clay, with very low permeability.

3.2.5 Geology and hydrogeology

The Karoo Supergroup is the largest and widespread stratigraphic unit in southern Africa. It is composed of the strata or sequence of strata that does not originate from marine processes, and it was deposited during the Late Carboniferous and Early Jurassic period of approximately 120 million years ago due to the tearing of South Africa from South America (Ahiakwo et al., 2018). Regarding its extent, the Karoo Supergroup covers about two-thirds of the surface of the land in southern Africa, for instance, it occurs in the Free State, Lesotho, and huge parts of the Northern Cape, Eastern Cape, KwaZulu-Natal and Mpumalanga provinces.

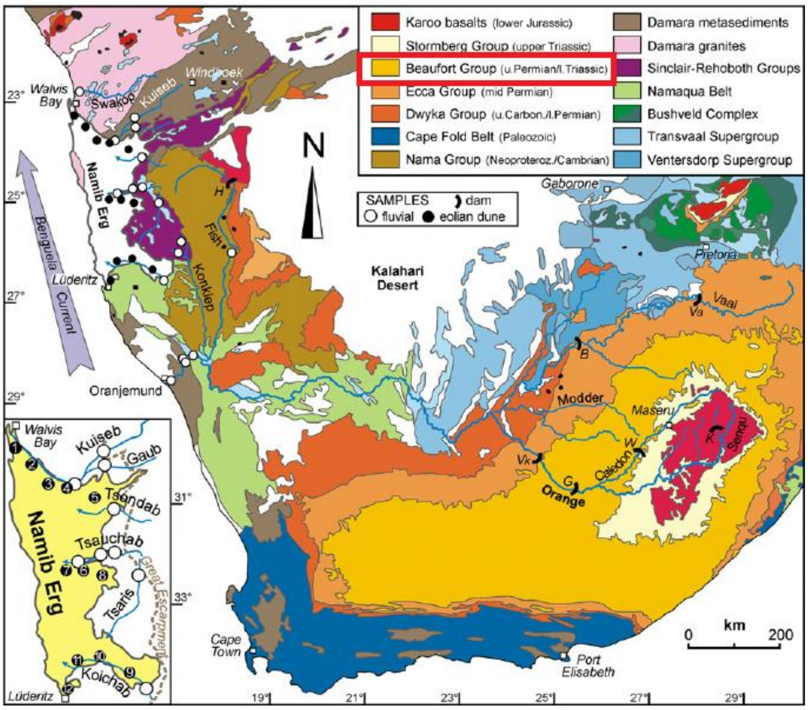


Figure 3.3 ***Geology of South Africa – Map outlining the Beaufort group, where the Botshabelo landfill is located*** *(Adopted from Garzanti et al., 2012)*

The creation of the Karoo Supergroup began by the break-up of the supercontinent Gondwana around 500 million years ago due to rifting processes to form a rift valley (Shone and Booth, 2005). This was followed by a large deposition of 8 km thick Cape Supergroup sediments that settled at the bottom of the valley, which filled the valley.

The two major tectonic forces that played an important role in the development of the Karoo Supergroup are the subduction zone along the southern boundary of Gondwana and rifting along the north. Cateneanu et al. (2005) found typical processes, including accretion, mountain construction, and extension or spreading in the north. As a result of the orogenic processes and the closing of the rift valley, folding and subduction continued to take place to form a sag basin in the south by mountain formation, creating a retroact foreland basin (Ahiakwo et al., 2018). Following the development of the basin, the Karoo Sea formed, and sedimentation occurred, resulting in the Karoo Supergroup. However, there was a general transition from cold to warmer and relatively hot climes, with corresponding changes in precipitation.

The Karoo Supergroup was deposited throughout the basin in a sequential fashion, forming significant lithostratigraphic groupings with diverse stratum compositions. The groups are: Dwyka, Ecca, Beaufort, Stormberg, and Drakensberg. The deposition of the Karoo Supergroup began with the Dwyka group at the bottom of the succession. It was influenced by the glacial processes due to the movement of ice sheets from north to south, and led to the formation of glacial valleys, thus, signifying the southward dipping elevation (Visser, 1993). During this movement, the deposition of diamictites, conglomerates, debris flow and sandstones were evident. As the ice sheets began to melt, there was a shift from the glacial to fluvioglacial processes that were triggered by changes in climatical conditions from cold to warmer, therefore leading to the deposition of the Ecca group (Johnson et al., 1996). The 3 000 m thick Ecca group is comprised of dark-coloured shales, siltstones and sandstones as well as coal seams that were deposited under reducing conditions of deltaic and fluvial conditions. As climates became warmer, the rainfall became seasonal due to semi-arid conditions that were formed. Cateneanu et al. (2005) mentioned that the term Beaufort typically refers to fluvial deposited rocks. The Beaufort group encompasses 20% of South Africa's total land surface and has a maximum thickness of 7 000 m. This group's rock types are primarily siltstones and mudstones, with sandstone lenses impacted by river systems. During the deposition of the Stormberg group, temperatures became exceedingly arid, exhibiting desert-like conditions typical of the Clarens formation, with loess type deposits and cross-bedded sand dunes (Johnson et al.,1996). However, the Elliot formation in the Stormberg group has red-coloured sandstones with upward fining in both sandstones and mudstones in a meandering river of low energy. Then the Drakensberg basalts erupted at the final stages of the Karoo Supergroup as Gondwana broke up. The Drakensberg group was formed, with the maximum thickness of 1 300 m, typically showing fissure-type of eruptions and lava flows that are associated with dolerite sills and dykes (Johnson et al., 1996).

Two aquifer types occur in the Mangaung Metropolitan Municipality, which are mainly fractured and intergranular aquifers. A fractured aquifer typically has water contained within fractures and joints, also including bedding planes and or contact zones between two strata (Mangaung Metropolitan Municipality, 2014). In the Beaufort group, groundwater occurrence is shown by Adelaide as well as the Tarkastad subgroups and is typical of dolerite intrusions. The average borehole yields are <0,5 ℓ/s, whereas in the dolerite dykes in between 0,5 and 5 ℓ/s (DWAF, 1996). The stratigraphy of the Karoo Supergroup, particularly in the Beaufort group, shows that there is a low permeability due to siltstones and mudstones, but the presence of fractures formed by dolerite dykes and sediment contact zones gives the aquifers high groundwater yielding capacity (Murray et al., 2012). This secondary porosity can be caused by processes such as weathering, brittle deformation, minor folding and faulting as well as jointing.

3.3 Waste management trends in the Mangaung Metropolitan Municipality

According to the Mangaung Metropolitan Municipality (2003), the Free State province generates about 80,2 million tons of waste per annum, in which 450 000 tonnes belongs to the municipality. The population of Mangaung approximates about 28% of the total population in the province, whereby, in 2011−2019 the population grew quantitatively from 775 028 to 878 834, which is a growth of 103 806 people (Mangaung Metropolitan Municipality, 2022). This shows a 1,6% growth rate in the population. The Botshabelo township proves to be the biggest township development in the Free State, formed in the year 1980, and experienced rapid population growth; however, it is to a large extent natural in nature with slow development in infrastructure. The area is predominantly a low-income housing in nature with 18 residential areas that are composed of blocks that are named by letters of the alphabet (Mangaung Metropolitan Municipality, 2020b). In the Mangaung region the sources of growth in the economy are transport, government, wholesale, retail and trade services

(Mangaung Metropolitan Municipality, 2016). The Botshabelo township has primary manufacturing factories that contribute to the local economy, despite its modest development. With rising population rates, the amount of waste generated rises as well, with research demonstrating that waste increase is directly related to wealth and economic levels. Out of the four dump sites in the municipality, one is located in the Botshabelo region as shown in Figure 3.4.

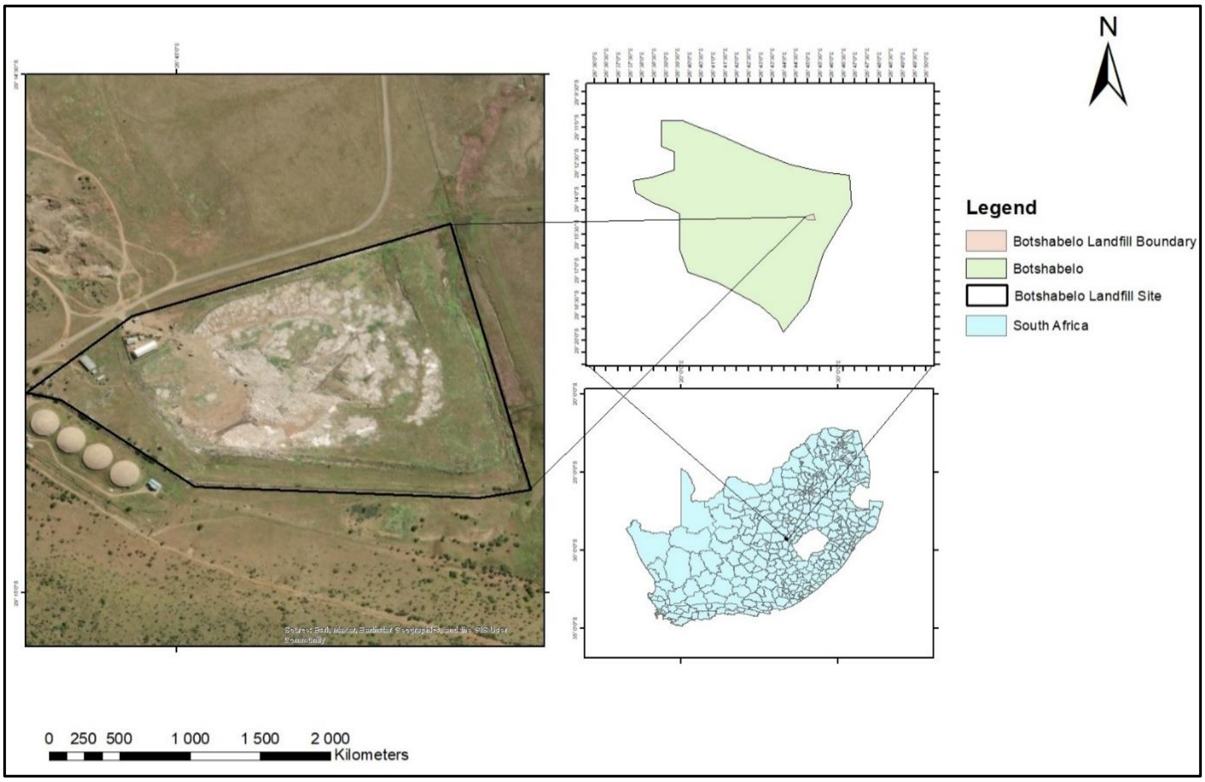


Figure 3.4 Map of Botshabelo *(Designed by author using ArcGIS 10 version)*

3.4 Site description

The Botshabelo landfill site is situated on the eastern side of Botshabelo (Mangaung Metropolitan Municipality, 2022). The site covers an area of around 15 ha and it was permitted according to Section 20 of the Environment Conservation Act 50 of 1997(Mangaung Metropolitan Municipality, 2022). The site receives non-hazardous or general type of waste from the surrounding areas that are mostly residential and commercial waste (Mangaung Metropolitan Municipality, 2014). In addition, the landfill site is designated as a General, Medium, Class B landfill (GMB), with a life period of 97 years based on the assessment done by the smec company in 2019. Typically, the general waste types that are dumped in the landfill include garden waste, domestic, construction, and tyres that are found disposed randomly across the landfill. The land in which the landfill site is situated is for agricultural purposes, but needs rezoning in order to accommodate the landfill. The Mangaung Metropolitan Municipality IWMP (2006) mentioned the presence of two monitoring boreholes that are located near the landfill site with periodical analyses. Although the landfill operation occurs on a regular basis, this does not mean that there are no issues on the site. For example, the site is not cutting-edge in terms of facilities due to frequent machinery breakdowns and inadequate management. These difficulties were common to the waste management sector in the Mangaung Metropolitan Municipality (2003).

3.5 Study design

The focus of the study in this section is on the phased approach in the achievement of the objectives. The study was done in seven phases, which included: assessment of study area, outlining of sampling points, sample collection, sample analysis, data analysis, geochemical speciation, vulnerability studies, and data presentation. Figure 3.5 is a flow diagram outlining the actual steps taken for the methodology.

3.5.1 Phase 1: Study area assessment

Figure 3.4 depicts the research region, the Botshabelo dump site, which is located on the boundaries of the Botshabelo Township. The phase assessment of the area is describing the area in greater detail to give a better perspective of the landfill in its totality. The tabulation of the descriptive summary is also provided in Table 3.1 for the sampling points delineated.

- **Botshabelo landfill site**

Location: The dump is located on the eastern side of the Botshabelo township (Figure 3.6). It is located in the level area next to the Pirana Mountain. It is located 1,3 km away from the settlement area, and thus, it is not visible because it is hidden by the mountains. It is located next to the abattoir. The Google map's location of the site is 29°14'46" S, 26°45'03" E.

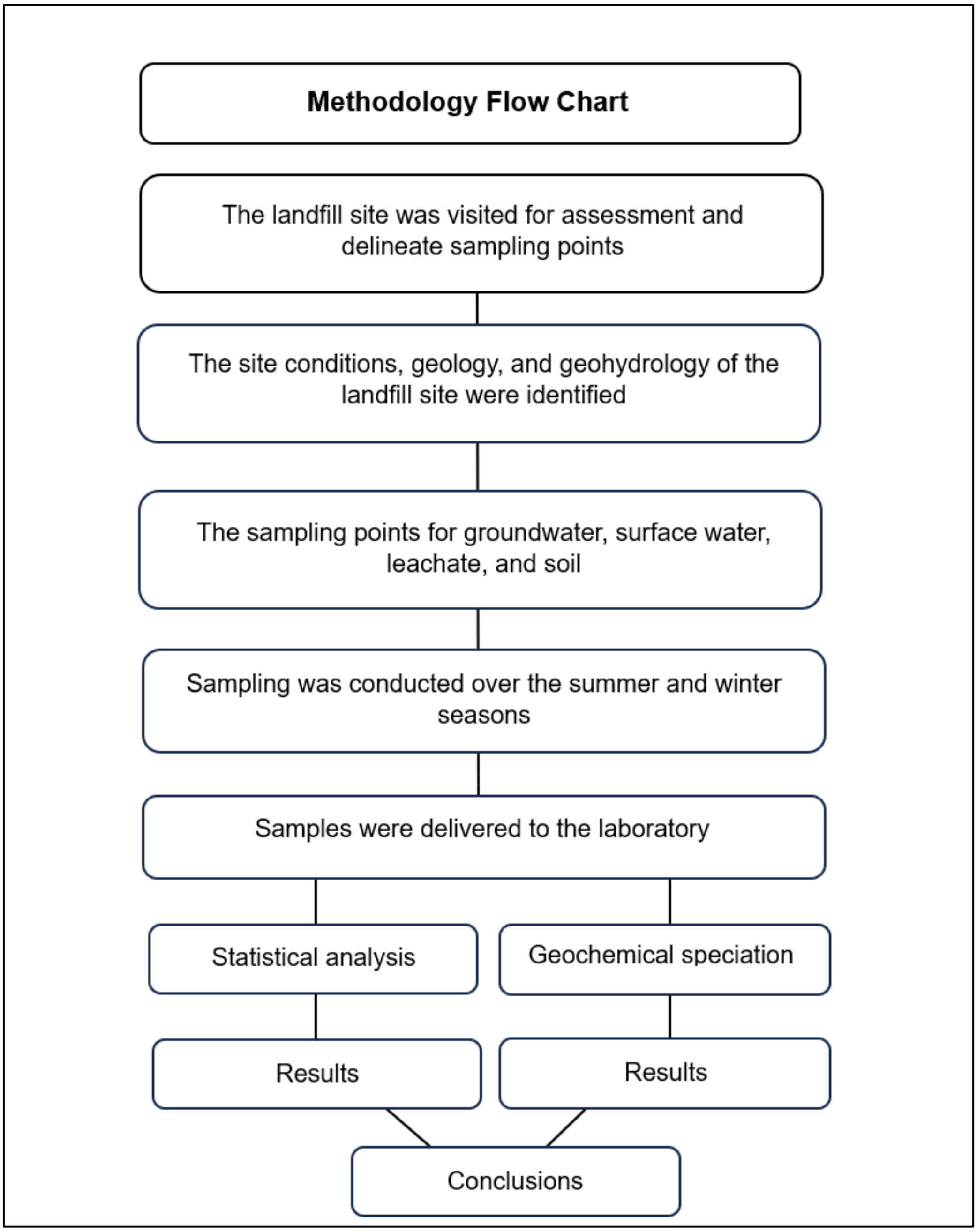


Figure 3.5 Methodology flow chart *(Designed by author)*

- **Study area geology**

The research region is situated in the Beaufort group of the Karoo Supergroup. It is composed of mudstone, sandstone, and shale sedimentary layers. Dolerite dykes cut through the rocks as intrusions occur, as evidenced by the ridge-like structures on the surface near the research region. Dolerites are common rock types found in the area (Figure 3.6).

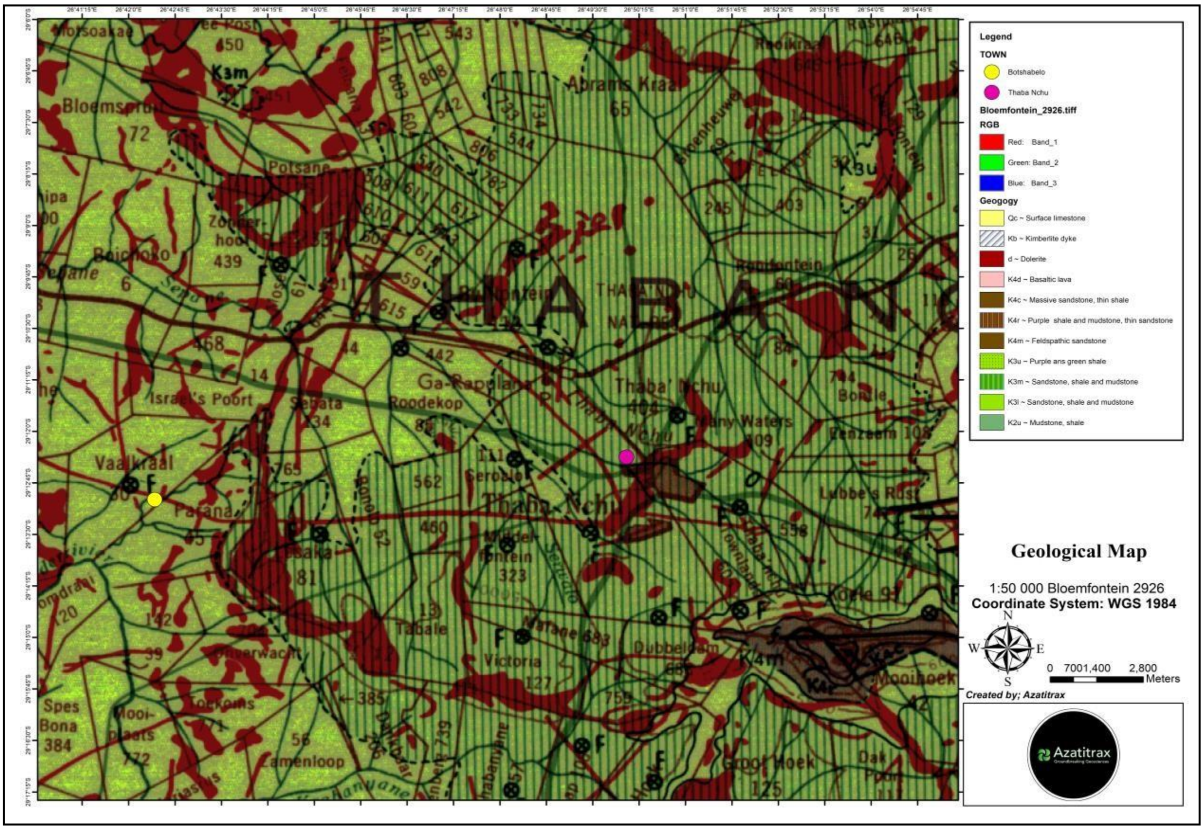


Figure 3.6 Geology of the Botshabelo landfill site *(Designed by the author using ArcGIS)*

- **Site condition**

The site is poorly managed as the infrastructure is not well developed, with trucks and other machinery broken down without being repaired. Waste tonnage measurement that measures the amount of waste daily is also not operational. During the day, there are waste pickers that normally carry out recycling work, with very few (at least two) trucks working on the site. The site officials are not working that much. Also, waste is scattered in and around the site and the site itself does not meet the type of a sanitary requirements. It is not a well-designed landfill that has a leachate collection site, waste compaction, as well as proper fencing. Open dumping is the main type of dumping on the site as depicted by Figure 3.7. One good thing about the site is the security access control system into the landfill that is well organised.

Figure 3.7 State of the Botshabelo landfill site *(Captured by the author)*

- **Type and amount of waste received**: The landfill site receives predominantly domestic types of waste as classified by the Mangaung Metropolitan Municipality (2011) as a general non-hazardous type of landfill. Furthermore, the landfill is classified as GMB (Mangaung Metropolitan Municipality, 2022). The general waste deposited at the landfill includes home debris, tyres, building rubble, and garden waste. The average type of waste received at the site between 2012 and 2019 was about 747 tons per month (Mangaung Metropolitan Municipality, 2022).
- **Identification of sampling points**: The site has a total of four monitoring boreholes that are located in singles (no duplicates) around the landfill. The surface water stream is also present in the proximity of the landfill. Table 3.1 depicts a descriptive summary of all the sampling points in the landfill site.

Table 3.1 Delineation of sampling points

Sampling point code	Coordinates	Sample type	Description	Distinguishing features
BH1*	29°14’41,02”S, 26°45’10,23” E	Borehole	The borehole is sealed with an iron cover. It is positioned on the northern side of the dump, which is closer to the landfill in distance. It is adjacent to the fence.	It is not pronounced, less visible and covered by vegetation and plastics. The collar is not visible. It has a 4,8 m depth.

Sampling point code	Coordinates	Sample type	Description	Distinguishing features
BH2	29°14’37,09”S, 26°45’13,99” E	Borehole	The borehole is located outside the landfill on its north-eastern side. There was no cap identified.	The collar is visible and elongated. It is 4,1 m deep from the surface to the water table.
BH3	29°14’ 42,31” S, 26°45’14,9” E	Borehole	The borehole is similar to the first borehole (BH1). It has a metal top and is placed on the landfill’s eastern edge. It is far from the landfill.	It is covered by vegetation and the collar is not visible. Its depth is 3,1 m from the surface, making it the shallowest borehole.
BH4	29°14’52,07” S, 26 45’19,34” E	Borehole	The bore is open with no cap inserted. It is much pronounced with clear visibility on the surface like borehole BH2. This borehole is the farthest from the landfill located on the south-eastern side.	It has a visible collar, with no vegetation cover. It is 10,37 m deep from the surface; the deepest borehole in the landfill site.
SW**	SW downstream 29°14’ 14,64”S, 26°45’ 20,68” E SW upstream 29°14’ 55,5” S, 26°45’ 19,34” E	Surface water	The creek runs on the eastern edge of the waste site and spills across boreholes BH1, BH2 and BH3 in a few metres. The stream is not a natural stream but artificial with the source the municipal reservoirs located upstream on the highland/mountainous feature adjacent to the landfill.	It is a runoff feature upstream, but it forms a standing water body due to the land surface at the middle of the stream. The stream flows far away from the landfill.
SS***	SS inside 29°14’ 46,244’’ S, 26°45’ 12,793” E SS outside 29°14’ 38,509” S, 26°45’ 14,181” E	Soil sample	The soil samples are largely located on the eastern side of the landfill, both inside and outside the landfill site.	The soils on both locations are normally black in colour- delineating the clay nature.
Leachate	29°14’ 46,809” S, 26°45’ 8,373” E	Leachate	The leachate is located in the centre-most part of the landfill where the waste pile is located.	It is shown by its dark brown to black coloured nature.

* BH = Borehole
** SW = Surface water
*** SS = Soil sample

Figure 3.8 shows the sampling points for boreholes and a surface water stream where water collection was conducted around the landfill site.

A – A capped borehole at the landfill site surrounded by plastic waste

B – The open borehole outside the landfill which has a pronounced collar

C – Borehole inside the landfill on the margin of the landfill located close to a pole for easy identification

D – Stream flow that passes through the landfill

Figure 3.8 Sampling points taken from the landfill

3.5.2 Phase 2: Sample collection

The sample collection phase included site visits to collect samples from the designated sampling stations in the dump. Water from boreholes, surface water, and leachate were collected in duplicate from each station. The landfill includes a total of four monitoring boreholes located both within and outside it. Samples were collected during two seasons, summer and winter, to investigate seasonal differences in water quality caused by leachate contamination. The first sampling was conducted in summer on 7 February 2023, which was marked by high summer temperatures and rainfall. On the other hand, in the winter, sampling was done on 30 July 2023, marking the final season with low winter temperatures and rainfall. Surface water was collected from its source that is located upstream, middle stream as well as downstream close to the landfill.

This was done to determine the quality of surface water at the source as compared to water quality in the middle and downstream near the dump. Similar to the work undertaken by Makhadi et al. (2020) and Ololade et al. (2019) at the Bloemfontein solid waste municipal sites, purging of the boreholes was not performed, which comprised the removal of pre-existing water to stabilise the field chemical parameters pH, DO, turbidity, and conductivity. This was because there was not enough data regarding the stabilization levels purging would attain. As shown in Table 3.1 the boreholes had varying depths, averaging 5,6 m.

- **Sampling approach for groundwater**

This study employed the sampling approach offered by the Institute for Groundwater Studies (IGS) at the University of the Free State to determine physical and microbiological parameters. The Central University of Technology and the IGS provided sampling equipment, which included a plastic bailer and water collection bottles (plastic for physicochemical parameters). The following are the step-by-step sampling procedures as set by IGS for physicochemical and biological parameters.

The landfill manager accepted the commencement of the sampling in the form of a written letter from the Department of Civil Engineering in the Groundwater division.

- **Sampling approach for physicochemical parameters**

  - Personal protective equipment was essential for sampling to take place safely. The first step for sampling was the wearing of gloves to avoid contamination of the samples. See Figure 3a-e for the sampling process.
  - This was followed by the opening of the borehole cap and insertion of the bailer into the borehole. As the bailer was submerged into the hole, upon filling the bailer, it was properly rinsed three times, repetitively.
  - The next step was to rinse the plastic bottles three times by using the borehole water. This was then followed by the actual collection of the borehole water that was filled into plastic bottles of 500 mℓ (Millilitre).
  - The bottles were filled adequately without overflowing and were therefore closed tightly to avoid contamination and to prevent air bubbles from entering.

- The bottles were then marked using a permanent marker for the correct codes of the borehole number and the date as well as correct location of the sampling point. They were then put inside the cooler box.
- This process was then followed from one borehole to the next. However, the bailer was rinsed thoroughly with the water of the borehole to avoid cross-contamination.

A – *Filling the plastic bottle with groundwater using bailer for physicochemical parameters*

B – *The bailer is inserted into the borehole for rinsing three times repeatedly*

C – *Surface water procedure for water collection using a plastic cup*

D – *Borehole water stored in cooler box in plastic bottles for physicochemical parameters*

E – *Surface water collected from the running stream around the landfill site using plastic cup*

Figure 3.9 Sampling data collection

- **Sampling approach for surface water**

The sampling method used in surface water was very different in that no bailer was used for water collection but only bottles were used to extract the water using a plastic cup that was sterile. This was conducted by physicochemical parameters.

- ***Sampling approach for soil***

Soil samples were collected at the landfill site using a shovel. Different locations were sampled, which included inside and outside the landfill. The collected soil was then placed inside small plastic bags that were sealed after infilling.

Soil collection was done using a shovel

The soil was put inside the plastic containers and sealed

Figure 3.10 Sampling of soil

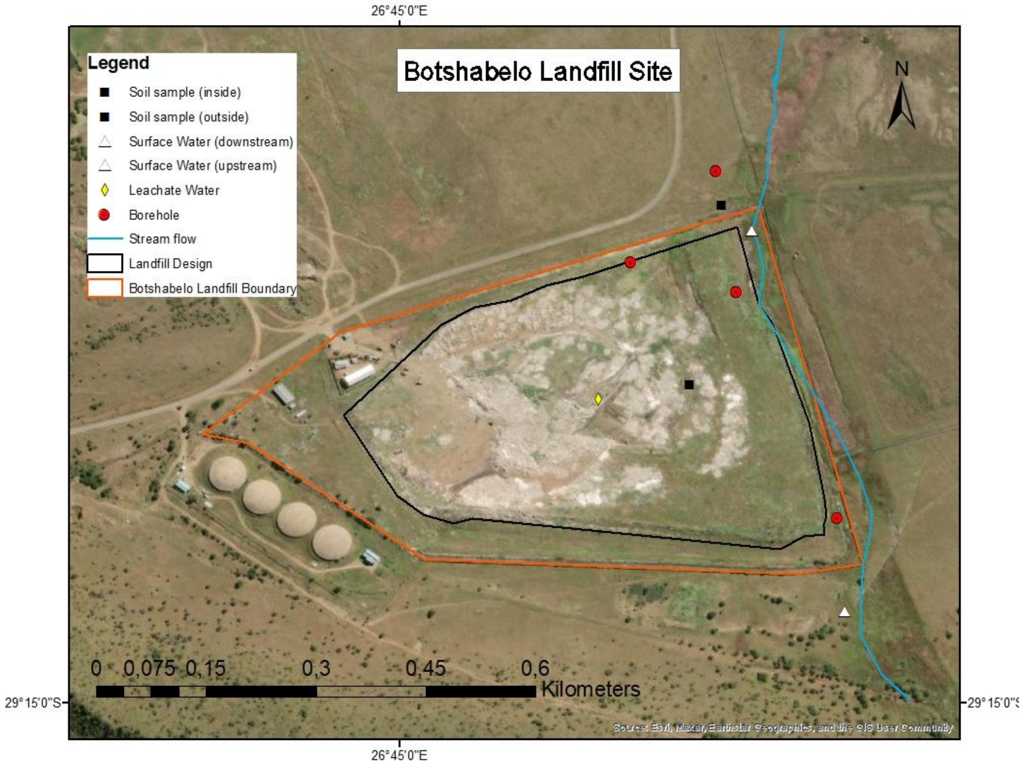


Figure 3.11 Botshabelo landfill site showing all sampling points: Landfill boundary, design, and stream

3.5.3 Sample analysis

Water quality is the physical, chemical, and biological properties of water that make it suitable for various uses (DWAF, 1996). These properties are determined by the dissolved or suspended elements of water. The obtained samples were then transferred to IGS for analysis. The IGS is a SANAS-accredited department that conducts water quality testing on many parameters. Heavy metals (Fe, Mn, Cr, and Pb) and main ions (Ca, Mg, Na, and K) were evaluated in the laboratory using spectrometers and a discrete analyser.

3.5.4 Physical and chemical parameters analysed

Table 3.2 shows the physicochemical parameters that were analysed using various methodologies. Heavy metals were assessed using Inductively Coupled Plasma (ICP), Inductively Coupled Plasma Optical Emission Spectroscopy (ICP−OES), and ICP−MS. Alkalinity, chloride, and nitrates were evaluated using a discrete analyser. The fluoride concentration was evaluated using the spectrophotometric technique. Nine ions (major cations and anions) were measured in milligrams per litre (mg/l). Metals were measured using SI units of mg/ℓ. This study assessed a total of twenty

(20) trace elements, nine (9) of which were heavy metals. The metals and heavy metals were both measured using the Teledyne Leeman Prodigy 7 ICP−OES similar to the study conducted by Makhadi et al. (2018) in the assessment of two landfill sites in the Bloemfontein city. Mg and Ca hardness were mathematically derived (Total Hardness [TH]).

3.5.5 Sample preparation for water samples

- **ICP−MS Spectrometers (ICP−MS and ICP−OES)**

To prepare the samples, 10 mℓ of each sample was filtered to eliminate any insoluble particles, and then a drop of nitric acid was added for digestion to break down the sample and liberate the analytes of interest. The parameters studied were Ca, Mg, Na, K, SO_4, Al, Ba, Boron (B), Mn, Cu, Ni, Fe Si, As, Antimony (Sb), Co, Cr, Cd, Molybdenum (Mo), Selenium (Se), Uranium (U), Zn, Pb, and Vanadium (V).

- **Discreet analyser**

This technique is highly automated and samples prepared are diluted, filtrated and put into the machine for analysis. Parameters that were analysed included alkalinity, Cl, NO_3^- and Nitrite (NO_2^-).

- **Spectrophotometry**

The parameter was determined with the technique by warming it for at least 15 minutes, followed by loading the samples into the cuvette with a maximum volume of 1 mℓ into which water samples were poured and then put into the technique. Only Flouride (F) was measured using this method.

- **Multiparameter waterproof meter (HI98195) and electrode meter (HI7698194-1)**

The Multiparameter waterproof meter was used to measure EC and TDS. The HI98195 was connected with an electrode probe sensor that was immersed into water that sent parameter signals to a display screen. The HI98195 was first calibrated to provide accurate measures of parameters. The HI7698194-1 electrode meter was used for measuring DO within the same method as the multiparameter waterproof meter HI98195.

- **In-situ pH**

The in-situ analysis was conducted when measuring the water pH from the study area using test strips. Analysis was done by pouring the water inside the container/cup. Then this was followed by putting the stripes inside the water, and this was repeated three times. Upon the colour change, a pH test range was used to compare with the colour on the strip.

3.5.6 Preparation of soil samples

- **Rigaku primus IV WDXRF**

The reference materials used for all calibrations were: BHVO-1, DR-N, GSP-1, JB-1, JB-2, JB-3, JG-1, JG-1A, JG-2, JG-3, JGB-1, NIM-D, NIM-G, SARM48, and SARM50. The importance of calibrations ensures the accuracy and optimum operation of the instrument by converting raw data signals into quantitative concentrations. Pressed pellets (used to measure Na and trace elements) from soil samples were prepared for the trace element analyses. A total of 8 g of the sample was added to 3 g of Hoechst wax ($C_6H_8O_3N_2$). This was then mixed for 20 minutes in a Turbula mixer. The mixture was then pressed to pressures greater than 395 newton per meter (N/m). The various measuring conditions for each specific trace element were used.

- **Five Go F2 portable pH/mV meters from Mettler-Toledo**

For pH readings, a 20 grams (g) portion of the sample was added to a 20 mℓ portion of deionised water. Samples were stirred and left to rest for an hour before pH readings were performed.

- **Malvern Panalytical Empyrean XRD**

Mineralogy analysis was used in conjunction with a Cu-anode X-ray tube to conduct XRD analysis. The high score was used for interpretation, together with the database ICDD-PDF2 2021. The results were interpreted and were semi-quantitative.

Table 3.2 Description of the parameters that were analysed and selected

Samples	Physical parameters	Chemical parameters
Groundwater and surface water samples	Electrical conductivity (EC)	pH, TDS, DO, salinity, alkalinity Hardness: Magnesium hardness, Calcium hardness Ions: SO_4, Mg, Ca, K, NO_2, NO_3, Cl Metals: As, Al, Sb, Beryllium (Be), Cd, Co, Barium (Ba), Fe, Ni, Pb, Se, B, Cr, Si, U, V, Zn, F, Si, Cu, Mo, Mn
Soil samples		pH Trace metals: Cr, Co, Ni, Cu, Zn, Mo, As, U, V, Ba, Mineralogy

3.5.7 Data analysis and presentation

- **Physicochemical parameters**

Data presentation entailed tabulating physicochemical parameters, which were then evaluated and compared to the WHO (2015) and SANS 241 (2015) drinking water quality standards. It was also compared to the DWAF (1996) guidelines for irrigation and domestic water use.

- **Analysis of dataset**

Microsoft Excel was used to generate a statistical variation between each parameter. Though the samples were collected in duplicates, the laboratory analysed only single samples for each sampling point due to cost constraints as numerous parameters had to be analysed. Therefore, no mean or average values were found for each parameter, but the dataset is deemed reliable for each sampling station. The analysed data were then used to compare the seasonal variations by the use of the coefficient of variation in order to compare how the parameters (physicochemical and biological) differed. This was performed by the utilisation of statistical analysis system software to determine the Analysis of Variance (ANOVA). The values were then compared using the least significant difference test at the 5% probability level with representative markings for each value.

- **Geochemical analysis**

The geochemical speciation was performed using Grapher Golden Software to analyse various species that occurred in the water samples. Major ion concentrations were used as inputs into the algorithm to detect major phases and hence classify the type of water in the samples. Metals had extremely low concentrations, thus they were not used. Piper, Stiff, Durov, and salinity hazard diagrams were utilised to categorise the principal ions, water type, geochemical evolution, and salinity hazard found in groundwater and surface water samples.

- **Salinity hazard plot**

SAR was used to determine the amount of salt in water as well as the salinity danger, which affected the usability of water for irrigation. The formula for calculating the SAR was as follows:

$$SAR = Na^{+} \div \frac{\sqrt{Ca^{2+} + Mg^{2+}}}{2} \qquad \textit{Equation 3.1}$$

The Ca, Na, and Mg cations represent their respective concentrations in water.

- **Pollution indices**

As noted in Chapter 2, the water and soil quality indices indicate the overall state of the water. They simplify complex datasets by reducing them to simple numerical values. These results are then used to gauge the quantity of contamination, using high or low values. Simple mathematical formulas are used for both physicochemical and biological parameters to give simple values. They are calculated by using parameters that have the most significant impact on contaminating water while those of least impact are not utilised. The presentation of data was mainly a tabulation of value.

- **Nemerow pollution index**

The NPI was calculated using the connection between a parameter's measured concentration and the background value (Zhang et al., 2018). The NPI was determined using this formula:

$$NPI = \frac{\mathrm{Ci}}{\mathrm{Li}}$$ *Equation 3.2*

Where
Ci represents the measured value of the ith parameter.
Li denotes the allowed limit of the ith parameter.

- **Drinking water quality index**

The Drinking Water Quality Index (DWQI) was used by the NSFWQI to determine the groundwater quality index (Geethamani et al., 2023). It was derived based on the weight assignment, relative weight (Wi), using Equation 3.3:

$$\mathrm{Wi} = \frac{\mathrm{Wi}}{\sum_{i=1}^{n} \mathrm{Wi}}$$ *Equation 3.3*

Where Wi is the weight of each unique water quality parameter and n is the total number of water quality parameters.

Individual weights were assigned to the water quality parameters based on their importance, with the most essential parameter receiving the greatest weight and the least important receiving the lowest.

Equation 3.4 yielded the following rating scale (qi) for each parameter:

$$\mathrm{qi} = \frac{\mathrm{Ci}}{\mathrm{Si}} \times 100$$ *Equation 3.4*

Where Ci represents the concentration of a single characteristic (i), unique units and Si represents the WHO water quality standard limit.

Equation 3.5 was computed for the sub-index (Sli) of each water quality parameter:

$$\mathrm{Sli} = \mathrm{Wi} \times \mathrm{qi}$$ *Equation 3.5*

Finally, the DWQI was determined using Equation 3.5, as follows:

$$DWQI = \sum \mathrm{Sli}$$ *Equation 3.6*

The WQI rating by DWQI is given in Table 3.3.

- **Weighted arithmetic water quality index**

The Weighted Arithmetic Water Quality index (WAWQI) was calculated to assess water quality using specified water quality measures (Brown et al., 1971). The WAWQI was calculated using the following formula:

$$WAWQI = \sum \mathrm{Qi\,Wi} \,/ \sum \mathrm{Wi} \qquad \textit{Equation 3.5}$$

The rating scale (Qi) for each individual parameter was then determined using the following formula:

$$\mathrm{Qi} = 100\left[\frac{\mathrm{Vi} - \mathrm{Vo}}{\mathrm{Si} - \mathrm{Vo}}\right] \qquad \textit{Equation 3.6}$$

Where Vi represents the concentration of the ith water parameter, while Vo represents its ideal value in pure water (Vo = 0). Si represents the recommended standard value for the ith parameter.

The following formula calculated the unit weight (Wi) for each unique parameter:

$$\mathrm{Wi} = \mathrm{K}/\mathrm{Si} \qquad \textit{Equation 3.7}$$

Where K is a constant that is obtained via the formula:

$$K = \frac{1}{\sum(\frac{1}{\mathrm{Si}})} \qquad \textit{Equation 3.8}$$

The rating of the WQI in accordance with WAWQI is given in Table 3.3.

Table 3.3 Classification of different water quality indices

Index method	Range	Classification	Application	References
NPI	≤1	No pollution		Su et al. (2022) Zhang et al. (2018)
	1,2	Slight pollution		
	2,3	Light pollution		
	3,5	Moderately polluted		
	>5	Seriously polluted		
WAWQI	0−25	Excellent water	Drinking, irrigation, industrial	Brown et al. (1970)
	25−50	Good water	Drinking, irrigation, industrial	
	50−75	Poor water	Irrigation, industrial	
	75−100	Very poor water	Irrigation	
	>100	Water unsuitable for drinking	Proper treatment before use	

Index method	Range	Classification	Application	References
NSFWQI	Below 50	Excellent		Geethamani et al. (2023)
	50−100	Good		
	101−200	Poor		
	201−300	Very poor		
	Above 300	Unsuitable for drinking		

- **Contamination factor**

The CF facilitates the evaluation of soil pollution by considering the trace element content of the soil as well as its background values, and was computed using the formula:

$$CF = C_{sample}/C_{background} \qquad \textit{Equation 3.9}$$

Whereby C_{sample} stands for concentration of individual trace element and $C_{background}$ represents permissible limits for trace element concentrations (Mugudamani et al., 2022).

The CF classification is illustrated in Table 3.4.

- **Geoaccumulation index**

The geoaccumulation index is one of the methods for calculating the contamination levels of trace elements in the soil medium. Developed by Muller (1969), it is applied to trace every individual trace element that posed pollution in the soil using the following formula:

$$\text{geoaccumulation index} = \log_2[\frac{\text{Cn}}{1{,}5\text{Bn}}] \qquad \textit{Equation 3.10}$$

Where Cn indicates concentration of heavy metals in soil and Bn is the trace metal's background value. Muller (1969) divided geoaccumulation index into seven classes as shown in Table 3.4.

- **Nemerow comprehensive index**

The Nemerow Comprehensive Index (NCI) was used to compute the total contamination of the soil without specifying individual characteristics that pose pollution (Nemerow, 1985). The pollution index by Nemerow was calculated in the following way:

$$NIPI = \sqrt{\mathrm{Plave}^2 + \mathrm{Plmax}^2} \qquad \textit{Equation 3.11}$$

Where $\frac{C_{heavymetal}}{C_{background}}$, Plave^2, and Plmax^2 represents the mean as well as the highest pollution index of every metal. The NCI is classified from no pollution to extremely severe pollution as depicted in Table 3.4.

Table 3.4 Classification of different soil quality indices

Index Method	Range	Classification	References
CF	<1	Low contamination	Devanesan et al. (2017), Jafaru et al. (2015)
	>1−3	Moderate contamination	
	>3−6	Considerably high contamination	
	>6	Very high contamination	
Geoaccumulation index	>5	Very strong	Jafaru et al. (2015)
	>4−5	Strong to very strong	
	>3−4	Strong	
	>2−3	Moderate to strong	
	>1−2	Moderate	
	>01	Practically uncontaminated to moderate	
	>0	Practically uncontaminated	
NCI	≤0,7	No pollution	Shan et al. (2022)
	0,7−1	Slight pollution	
	1−2	Moderate pollution	
	2−3	Severe pollution	
	>3	Extremely severe pollution	

- **Groundwater susceptibility/vulnerability**

To estimate the groundwater sensitivity to pollution, two models were utilised, primarily: GOD and RTt models. Each and every model's name represents factors or parameters as stated in Chapter 2, Similar to the WQI calculation, parameters were allocated a value based on its capacity to contaminate groundwater. Data presentation was done by the use of ArcGIS software to produce maps that show vulnerability of groundwater to contamination. On the maps were areas that had low, medium and high degree contamination stipulated by various colours.

- **GOD vulnerability index method**

This approach was developed to focus primarily on the vertical infiltration of pollutants in the unsaturated zone (Salim et al., 2019). It was calculated using the following formula:

$$GOD = G \times O \times D$$ *Equation 3.12*

Where G represents the groundwater occurrence, O denotes the overlaying lithology, and D denotes the aquifer depth. The GOD index was calculated in respect of the ratings of each parameter, which included numerous ranges, to produce a total score/weight of each individual parameter according to Table 3.5. In Table 3.5 the vulnerability classes are shown for the GOD index.

Table 3.5 Parameters of the GOD vulnerability index

God parameters	Range	Rating	Weight	Total weight (rating * weight
Groundwater Confinement	Overflowing	0	-	0
	Confined	0,2	-	0,2
	Semi-confined	0,6	-	0,6
	Unconfined	1,0	-	1,0
Overlying strata	Residual soil	0,4	-	0,4
	Limon alluvial, loess, shale, fine limestone	0,5	-	0,5
	Aeolian sand, siltiest, turf, igneous rock	0,6	-	0,6
	Sand and gravel, sandstone, tufa	0,7	-	0,7
	Gravel	0,8	-	0,8
Depth to water table (m)	5−10	0,8	-	0,8
	10−20	0,7		0,7
	20−50	0,6		0,6

- **Rainfall travel-time vulnerability method**

The formulation of this method involved the rating of the parameters that critically affect the overall vulnerability of groundwater (Oke et al., 2016). The components of the RTt method included R-factor, which is the recharge as a result of rainfall input that directly impacts the contaminant. The following expression was deduced for the rainfall factor:

$$R \times 10$$ *Equation 3.13*

Where, R is the rainfall of a specific area.

The overall depth and slope also played a role in the vulnerability and were calculated as follows:

$$D \times S$$ *Equation 3.14*

D stands for the depth and S for slope.

The Tt factor is the rate of flow that was assumed according to the equation:

$$Tt = D \times S/(\frac{\mathrm{Ksat}}{\theta})$$ *Equation 3.15*

In which, $\frac{\mathrm{Ksat}}{\theta}$ is the pore velocity divided by the porosity of the source and receptor, respectively.

The final RTt vulnerability was then computed using the formula below:

$$Tt = R + (10) + D \times S/(\frac{\mathrm{Ksat}}{\theta}) \times 10$$ *Equation 3.16*

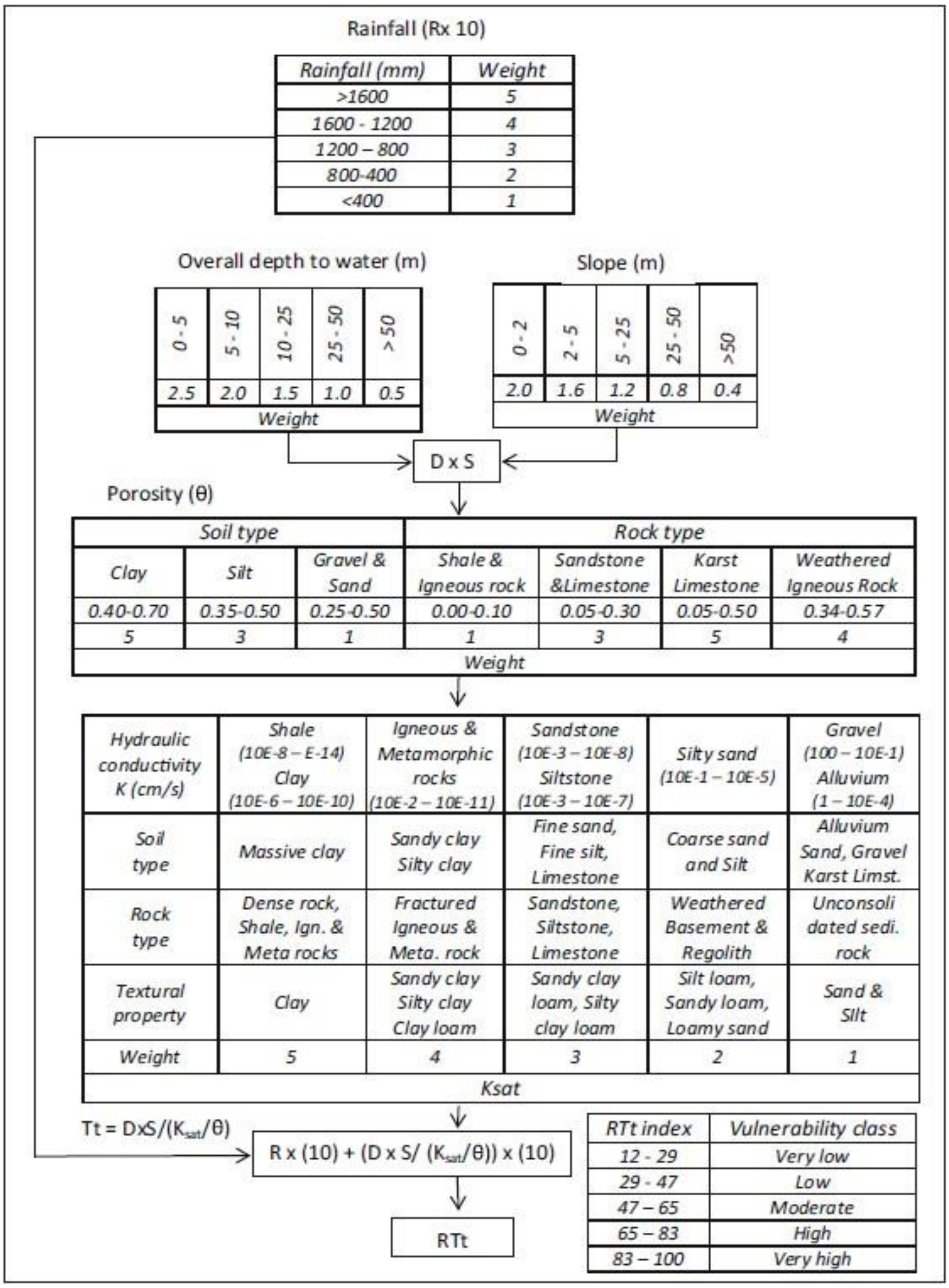
Figure 3.12 shows the step-by-step development of the RTt method, together with the weights assigned to every contributing parameter. Table 3.6 depicts the overall vulnerability classes for the RTt index.

Table 3.6 Classification of the GOD and RTt indices

Vulnerability method	Class	Index value
GOD	0−0,1	Negligible
	0,1−0,3	Low
	0,3−0,5	Moderate
	0,5−0,7	High
	0,7−1,0	Very high
RTt	12−29	Very low
	29−47	Low
	47−65	Moderate
	65−83	High

Vulnerability method	Class	Index value
	83−100	Very high

Figure 3.12 shows the development of the RTt method



Rainfall (Rx 10)

Rainfall (mm)	Weight
>1600	5
1600 - 1200	4
1200 – 800	3
800-400	2
<400	1

Overall depth to water (m)

0 - 5	5 - 10	10 - 25	25 - 50	> 50
2.5	2.0	1.5	1.0	0.5
Weight				

Slope (m)

0 - 2	2 - 5	5 - 25	25 - 50	>50
2.0	1.6	1.2	0.8	0.4
Weight				

Porosity (θ)

Soil type			Rock type			
Clay	Silt	Gravel & Sand	Shale & Igneous rock	Sandstone &Limestone	Karst Limestone	Weathered Igneous Rock
0.40-0.70	0.35-0.50	0.25-0.50	0.00-0.10	0.05-0.30	0.05-0.50	0.34-0.57
5	3	1	1	3	5	4
Weight						

Hydraulic conductivity K (cm/s)	Shale (10E-8 – E-14) Clay (10E-6 – 10E-10)	Igneous & Metamorphic rocks (10E-2 – 10E-11)	Sandstone (10E-3 – 10E-8) Siltstone (10E-3 – 10E-7)	Silty sand (10E-1 – 10E-5)	Gravel (100 – 10E-1) Alluvium (1 – 10E-4)
Soil type	Massive clay	Sandy clay Silty clay	Fine sand, Fine silt, Limestone	Coarse sand and Silt	Alluvium Sand, Gravel Karst Limst.
Rock type	Dense rock, Shale, Ign. & Meta rocks	Fractured Igneous & Meta. rock	Sandstone, Siltstone, Limestone	Weathered Basement & Regolith	Unconsoli dated sedi. rock
Textural property	Clay	Sandy clay Silty clay Clay loam	Sandy clay loam, Silty clay loam	Silt loam, Sandy loam, Loamy sand	Sand & Silt
Weight	5	4	3	2	1
Ksat					

$Tt = DxS/(K_{sat}/\theta)$

$R \times (10) + (D \times S/\ (K_{sat}/\theta)) \times (10)$

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

Figure 3.12 Rainfall travel-time method development *(Source: Oke, 2016)*

Chapter 4

Results and Discussion

4.1 Introduction

The chapter presents the laboratory results that were examined and reported. The results include the physicochemical parameters of the surface water and groundwater, leachate, as well as the soil. The discussion of the data entails the outline of the results in comparison with the predetermined standards in conjunction with the work of other researchers in the same line of context. The presentation of the data is given using tables, maps and figures of the chemistry, geochemistry, WQIs, and vulnerability studies. This chapter also presents the results of the water quality and suitability for various purposes, including drinking and irrigation. This chapter gives the water quality determination throughout both the rainy and dry seasons.

4.2 Water quality

4.2.1 Physicochemical parameters

The importance of physicochemical parameters is to assess the quality of water before it is used for drinking, residential, agricultural, and industrial applications (Dirican, 2015). Various physicochemical criteria are utilised to determine the water quality for specific usage in comparison to set standards that guide such use. Physicochemical parameters entail both physical and chemical elements that are found in water, where physical determinants entail EC. On the other hand, chemical determinants include pH, TDS, DO, major ions, trace elements, heavy metals, alkalinity, and total hardness.

4.2.2 Leachate characteristics

Table 4.1 compares the physicochemical properties of leachate to worldwide and national water quality criteria published by the WHO (2022), SANS 241 (2015), and DWAF (1996c) for irrigation.

Table 4.1 Physicochemical parameters of the leachate

Note: During the dry season there was no leachate generated due to little or no rainfall.

Parameters	Unit	Leachate	WHO	SANS 241	DWAF
Ca	mg/ℓ	64,90	75	–	–
Mg	mg/ℓ	58,85	30	–	–
Na	mg/ℓ	495,13	200	–	≤70
K	mg/ℓ	75,23	300	≤200	–
Cl	mg/ℓ	483,39	200-300	≤300	≤100
NO_3^-	mg/ℓ	<0,16	50	≤11	–
NO_2^-	mg/ℓ	1,91	3	≤0,9	–
SO_4	mg/ℓ	32,62	500	≤500	–
Alkalinity	mg/ℓ	647,02	–	–	–
Ca Hardness	mg/ℓ	162,05	–	–	–
Mg Hardness	mg/ℓ	242,36	–	–	–
TH	mg/ℓ	404,40	–	–	–
Al	mg/ℓ	0,10	≤0,1	≤300	≤5
As	mg/ℓ	0,01	0,01	≤10	≤0,1
Sb	mg/ℓ	<0,00	0,02	≤20	–
Ba	mg/ℓ	0,15	1,3	≤700	–
B	mg/ℓ	0,23	2,4	≤2400	≤0,5
Cd	mg/ℓ	<0,00	0,003	≤3	≤0,01
Co	mg/ℓ	0,084	–	–	≤0,05
Cr	mg/ℓ	0,09	0,05	≤50	≤0,1
Cu	mg/ℓ	0,18	2	≤2000	≤0,2
Fe	mg/ℓ	0,15	–	–	≤5
Mn	mg/ℓ	0,18	0,08	≤2000	≤0,02
Ni	mg/ℓ	0,21	0,07	≤400	≤0,20
Mo	mg/ℓ	0,01	–	≤70	≤0,01
Pb	mg/ℓ	<0,00	0,1	≤10	≤0,20
Se	mg/ℓ	<0,00	0,04	≤40	≤0,02
U	mg/ℓ	0,00	0,03	≤30	≤0,01
V	mg/ℓ	0,22	–	–	≤0,10
Zn	mg/ℓ	0,01	3	≤5	≤1
F	mg/ℓ	1,00	1,5	≤1,5	≤2

4.2.3 Alkalinity and total harness

The leachate had an alkalinity of 647,02 mg/ℓ, respectively. It should be noted that the leachate sample was only collected during the rainy season; during the dry season, there was little to no rainfall, resulting in little leachate formation. The high amounts of alkalinity were due to the biochemical breakdown process of organic matter occurring inside the landfill, which results in the creation of bicarbonate, a primary component of alkalinity (Naveen et al., 2016). Extreme alkalinity of the leachate can pose a significant risk to groundwater contamination and results in unsatisfactory odour in the groundwater. The Ca and Mg hardness of the leachate were 162,05 mg/ℓ and 242,36 mg/ℓ, respectively, which both had greater concentrations than all the water samples in both seasons. The overall total hardness of the leachate was 404,40 mg/ℓ, which was much greater than that of the groundwater and surface water throughout the year. According to Naveen et al. (2014), increased hardness is caused by the carbonate and bicarbonate of Ca and Mg. Ash and slag produced by the combustion of agricultural waste, wood, and peat are responsible for the high amounts of alkalinity and hardness. Not all of the water quality criteria employed in this study included levels for alkalinity and total hardness.

4.2.4 Major cations

The Ca and Mg concentrations in leachate were 64,90 mg/ℓ and 58,85 mg/ℓ, respectively, which were within the limits of the WHO drinking water quality standard; however, no limitations were stated for SANS and DWAF. The leachate included 495,00 mg/ℓ of Na, which was higher than the WHO and DWAF guidelines. Additionally, K had higher quantities than all water samples at 75,23 mg/ℓ, but it fell within the limits of the WHO and DWAF criteria. Na and K are two of the most prevalent components of the leachate that is transmitted from the trash through mass transfer operations as well as the stabilisation phase of the landfill, resulting in high salinity of leachates (Naveen et al., 2014). The sources of Na in leachates can be tiles, plaster and concrete that release Na by degradation.

4.2.5 Major anions

NO_3^- values of 0,16 mg/ℓ were found to be safe for and met the WHO requirements. This means that no agricultural waste was placed in the landfill site, including garbage from the abattoir next to it. NO_2^- was somewhat more enriched than NO_3^- at 1,91 mg/ℓ, exceeding the SANS criterion but less than the WHO guideline. Both NO_3^- and NO_2^- showed less enrichment in the leachate sample, and according to Omofunmi et al. (2020), the concentration of leachate fluctuates depending on the type of waste or its composition. To support this, the landfill itself is a general type of landfill that does not receive very hazardous type of waste. SO_4 did not depict excess enrichment as it had a concentration of 32,62 mg/ℓ. However, Cl had the highest concentration of all the major anions at 483,39 mg/ℓ, which exceeded both the SANS and WHO standards. Similar investigations indicated Cl enrichment at dump sites (Naveen et al., 2014, 2018; Omofunmi et al., 2020). The presence of soluble salts in the landfill site, as well as animal waste, sewage, and agricultural wastes, can all contribute to increased Cl concentrations. Furthermore, the excessive Cl levels could be produced by diapers, which are common in the research location.

4.2.6 Trace elements

The trace element concentrations of the leachate for Al, As, Sb, B, Cd, Fe, Mo, Pb, Se, U, Zn, F, Cu, and Cr showed very low concentrations that fell within the standard limits for the WHO, SANS, and DWAF. The recorded values in Table 4.1 were 0,10 mg/ℓ, 0,01 mg/ℓ, <0,01 mg/ℓ, 0,23 mg/ℓ, <0,00 mg/ℓ, 0,15 mg/ℓ, 0,01 mg/ℓ, <0,00 mg/ℓ, <0,00 mg/ℓ, 0,00 mg/ℓ, 0,01 mg/ℓ, 1,00 mg/ℓ, 0,18 mg/ℓ, and 0,09 mg/ℓ, respectively. However, the remaining trace elements showed pollution of the leachate by having concentrations exceeding the standards given in Table 4.1. Co, Mn, V(vii), and Ni had values at 0,08 mg/ℓ, 0,18 mg/ℓ, and 0,21 mg/ℓ, respectively. Landfills are the sources of heavy metal which originate from wastes such as ceramins, batteries, electronic products, light bulbs as well as glass (Naveen et al., 2018). The study by Przydatek and Kanownik (2019) showed low concentrations for heavy metals in correlation to this study. Furthermore, the Botshabelo landfill site has been in operation for more than 15 years, making it an old landfill, which may explain the general low enrichment of the leachate. In addition, leachates from historic landfills are high in alkalinity due to anaerobic waste decomposition (Lindamulla et al., 2022).

4.3 Groundwater and surface water quality

Table 4.2 shows the average physical and chemical composition of groundwater and surface water throughout both the wet and dry seasons. In this study, the physicochemical characteristics of groundwater and surface water were compared to international drinking water standards such as the WHO standard (2015), SANS (2015), and DWAF (1996) irrigation standards.

Table 4.2 The mean concentrations of physicochemical parameters of groundwater and surface water samples over wet and dry seasons

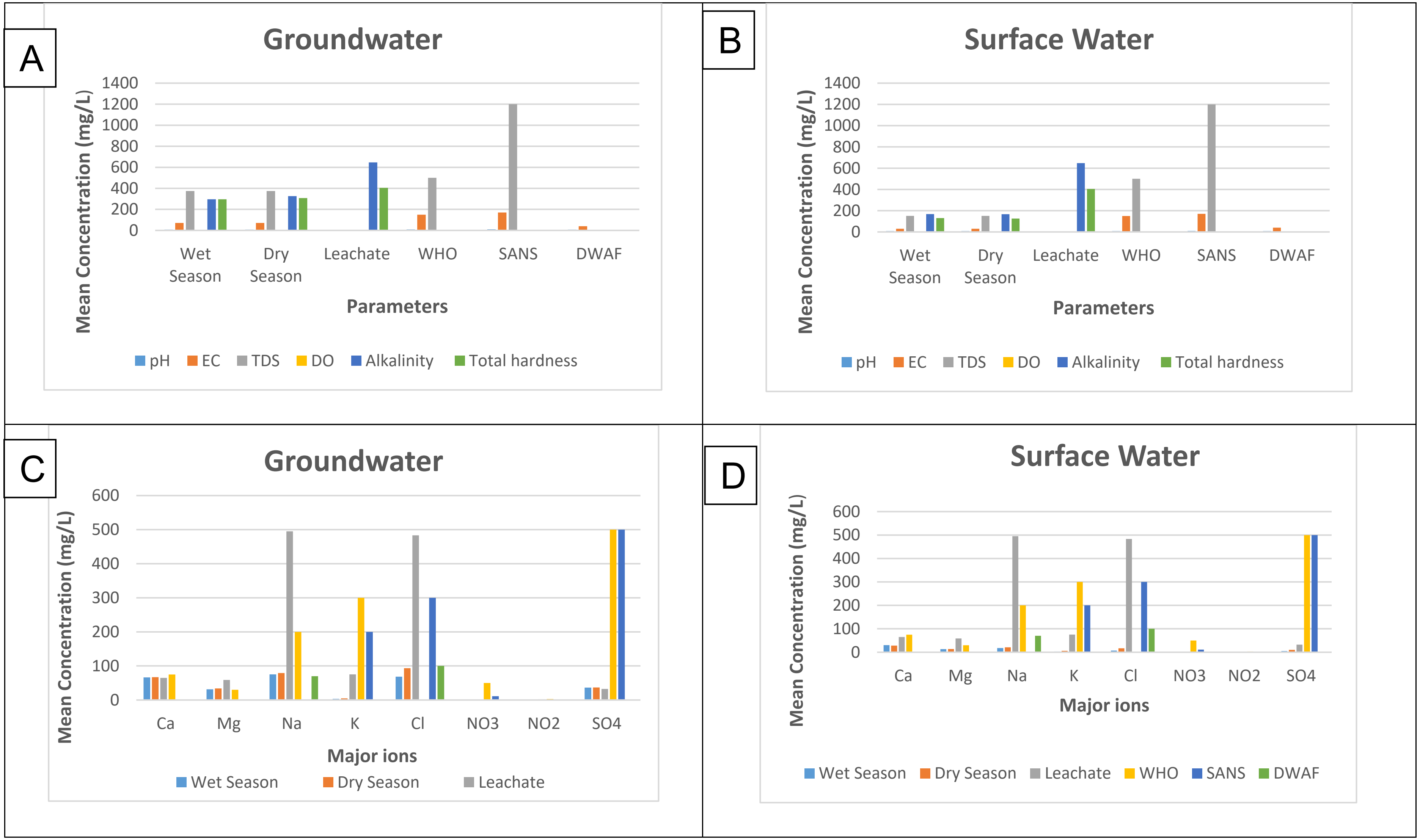
Parameters	Unit	Groundwater		Groundwater		Surface water		Surface water		National standards		
		Wet season		Dry season		Wet season		Dry season		WHO	SANS	DWAF
		Min-Max	Mean±SD	Min-Max	Mean±SD	Min-Max	Mean±SD	Min-Max	Mean±SD			
pH		8,00-8,00	8,00±0,00	8,00-8,00	8,00±0	8,00-8,00	8,00±0,00	8,00-8,00	8,00±0,00	6,5-8,5	5,0-9,7	6,5-8,4
EC	mS/m	44,80-95,70	71,20±18,46	44,80-93,60	69,20±34,5	27,50-32,70	30,30±3,39	27,90-33,50	30,70±3,95	150	≤170	≤40
TDS	mg/ℓ	300-482	375±79,67	305,0-482,0	393,5±125,16	140-163	151,10±16,26	139,0-167,0	153,0±19,8	500	≤1200	-
DO	mg/ℓ	3,01-6,22	4,38±518,27	3,00-6,20	4,60±2,26	3,17-3,72	3,44±0,38	3,20-3,72	3,46±0,37	-	-	-
Ca	mg/ℓ	46,13-91,39	66,26±21,17	43,76-97,70	67,09±25,04	30,2-30,45	30,325±0,176	28,03-28,47	28,25±0,31	75	-	-
Mg	mg/ℓ	25,59-36,84	31,50±4,66	30,66-38,42	34,00±3,57	12,44-14,20	13,32±1,244	13,51-13,62	13,565±0,07	30	-	-
Na	mg/ℓ	50,43-105,96	75,11±23,97	43,25-115,06	79,09±31,00	16,3-19,55	17,925±2,29	16,36-26,21	21,285±6,96	200	-	≤70
K	mg/ℓ	1,37-7,32	3,27±2,73	3,31-7,32	4,91±1,71	2,19-4,06	3,125±1,322	5,11-6,51	5,81±0,98	300	≤200	-
Cl^-	mg/ℓ	6,21-131,4	68,53±51,21	12,42-149,58	93,25±58,93	6,71-7,50	7,105±0,558	13,37-20,73	17,05±5,20	200-300	≤300	≤100
NO_3^-	mg/ℓ	0,16-0,16	0,16±00	0,16-1,02	0,39±0,42	0,16-0,16	0,16±0,00	0,16-0,16	0,16±0,00	50	≤11	-
NO_2^-	mg/ℓ	0,007-0,02	0,01±0,01	0,01-0,02	0,01±0,01	0,006-0,012	0,009±0,004	0,01-0,01	0,01±0,00	3	≤0.9	-
SO_4	mg/ℓ	18,63-67,02	36,51±22,82	14,35-66,11	36,63±21,79	4,7-5,24	4,97±0,38	7,4-12,91	10,155±3,89	500	≤500	-
Alkalinity	mg/ℓ	250,28-341,10	295,99±37,08	294,70-352,97	325,69±29,17	162,42-172,96	167,69±7,45	162,27-172,77	167,52±7,42	-	-	-
Ca hardness	mg/ℓ	115,19-228,20	160,45±50,72	109,47-243,96	167,56±62,45	75,49-76,04	75,765±0,39	69,98-72,09	70,535±0,78	-	-	-
Mg hardness	mg/ℓ	105,37-151,71	129,72±19,21	126,26-158,21	140,00±14,70	51,24-58,38	54,81±5,04	55,65-56,08	55,865±00,30	-	-	-
TH	mg/ℓ	234,25-379,91	295,17±66,85	239,34-402,17	307,51±71,08	127,28-133,88	130,58±4,66	125,63-127,18	126,405±1,09	-	-	-
Al	mg/ℓ	0,06-0,10	0,08±0,02	0,01-0,02	0,01±0,01	0,049-0,058	0,0535±0,006	0,02-0,13	0,075±0,07	<0.1	≤300	≤5
As	mg/ℓ	0,00-0,00	0,00±0,00	0,00-0,00	0,00±0,00	0,00-0,00	0,00±0,00	0,00-0,00	0±0,00	0,01	≤10	≤0,1
Sb	mg/ℓ	0,00-0,00	0,00±0,00	0,00-0,00	0,00±0,00	0,00-0,00	0±0	0-0	0±	0,02	≤20	-

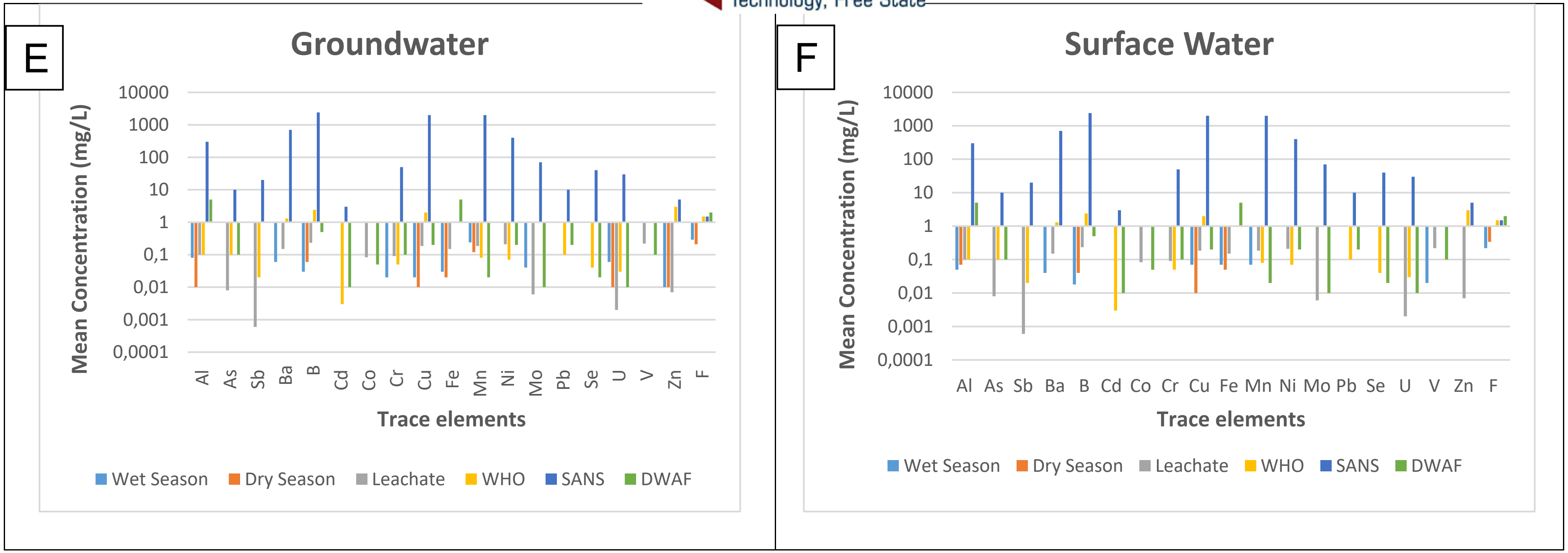
Parameters	Unit	Groundwater		Groundwater		Surface water		Surface water		National standards		
		Wet season		Dry season		Wet season		Dry season		WHO	SANS	DWAF
		Min-Max	Mean±SD	Min-Max	Mean±SD	Min-Max	Mean±SD	Min-Max	Mean±SD			
Ba	mg/ℓ	0,04-0,09	0,06±0,02	0,00-0,00	0,00±0,00	0,035-0,053	0,044±0,01	0-0	0±0	1,3	≤700	-
B	mg/ℓ	0,02-0,04	0,03±0,01	0,05-0,06	0,06±0,01	0,017-0,02	0,0185±0,002	0,04-0,4	0,04±0	2,4	≤ 2400	≤0.5
Cd	mg/ℓ	0,00-0,00	0,00±0,00	0,00-0,00	0,00±0,00	0-0	0±0	0-0	0±0	0,003	≤3	≤0.01
Co	mg/ℓ	0,00-0,00	0,00±0,00	0,00-0,00	0,00±0,00	0-0	0±0	0-0	0±0	-	-	≤0.05
Cr	mg/ℓ	0,01-0,02	0,02±0,01	0,00-0,00	0,00±0,00	0-0	0,0005±0,00	0,002-0,002	0,002±0	0,05	≤50	≤0.1
Cu	mg/ℓ	0,00-0,07	0,02±0,03	0,01-0,01	0,01±0,00	0,027-0,127	0,077±0,07	0,01-0,01	0,01±0	2	≤2000	≤0.2
Fe	mg/ℓ	0,00-0,07	0,03±0,03	0,01-0,05	0,02±0,02	0,05-0,102	0,076±0,03	0,01-0,1	0,055±0,06	-	-	≤5
Mn	mg/ℓ	0,00-0,86	0,24±0,41	0,00-0,46	0,12±0,22	0,027-0,127	0,077±0,07	0,005-0,01	0,0075±0,003	0,08	≤2000	≤0.02
Ni	mg/ℓ	0,00-0,00	0,00±0,00	0,00-0,00	0,00±0,00	0-0	0±0	0,002-0,003	0,0025±0,00	0,07	≤400	≤0.20
Mo	mg/ℓ	0,01-0,12	0,04±0,05	0,00-0,00	0,00±0,01	0-0	0±0	0,001-0,002	0,0015±0,00	-	≤70	≤0.01
Pb	mg/ℓ	0,00-0,00	0,00±0,00	0,00-0,00	0,00±0,00	0-0	0±0	0-0	0±0	0,1	≤10	≤0.2
Se	mg/ℓ	0,00-0,00	0,00±0,00	0,00-0,00	0,00±0,00	0-0	0±0	0-0	0±0	0,04	≤40	≤0.02
U	mg/ℓ	0,01-0,11	0,06±0,05	0,00-0,01	0,01±0,00	0	0±0	0,003	0,0015±0,003	0,03	≤30	≤0.01
V	mg/ℓ	0,00-0,00	0,00±0,00	0,00-0,01	0,00±0,00	0-0,047	0,0235±0,03	0,001-0,007	0,004±0,004	-		≤0.10
Zn	mg/ℓ	0,00-0,01	0,01±0,00	0,00-0,01	0,01±0,00	0,004-0,004	0,004±0	0,0010,001	0,001±0	3	≤5	≤1
F	mg/ℓ	0,25-0,38	0,29±0,06	0,16-0,28	0,21±0,05	0,21-0,23	0,22±0,01	0,2-0,48	0,34±0,19	1,5	≤1.5	≤2

SD= Standard deviation

4.3.1 pH

The average pH of groundwater and surface water in both the rainy and dry seasons was 8,00. These results confirmed that the groundwater and surface water in this study were alkaline and fell below the WHO, SANS, and DWAF limits (Figure 4.1a). These findings are consistent with other groundwater investigations conducted near dump sites (Abiriga et al., 2020; Ololade et al., 2019). Makhadi et al. (2020) detected an alkaline pH in surface water around the dump site. The results of alkaline water around the landfill site in this study may also suggest influx of alkaline leachate (Abiriga et al., 2020, Ravikumar (2015) also observed pH values ranging from 6,52 to 8,55 and 7,15 to 7,50 in both groundwater and surface water that were acidic to alkaline, thus indicating preferable drinking consumption. Dumping of ashes in the landfill site may also explain these outcomes. The fact that the majority of the waste received at the Botshabelo landfill site was household waste, rather than waste from industry or mining, which may have also contributed to the alkaline pH found in the water bodies surrounding the site. This area's alkaline surface and groundwater may indicate that the local population and aquatic life are unlikely to be negatively impacted by health issues. Low pH averages can harm plants when used for irrigation and cause gradual erosion of dental enamel in those who use the water (DWAF, 1996).


A
Groundwater
Mean Concentration (mg/L)
1400
1200
1000
800
600
400
200
0
Wet Season
Dry Season
Leachate
WHO
SANS
DWAF
Parameters
pH
EC
TDS
DO
Alkalinity
Total hardness
B
Surface Water
Mean Concentration (mg/L)
1400
1200
1000
800
600
400
200
0
Wet Season
Dry Season
Leachate
WHO
SANS
DWAF
Parameters
pH
EC
TDS
DO
Alkalinity
Total hardness
C
Groundwater
Mean Concentration (mg/L)
600
500
400
300
200
100
0
Ca
Mg
Na
K
Cl
NO3
NO2
SO4
Major ions
Wet Season
Dry Season
Leachate
D
Surface Water
Mean Concentration (mg/L)
600
500
400
300
200
100
0
Ca
Mg
Na
K
Cl
NO3
NO2
SO4
Major ions
Wet Season
Dry Season
Leachate
WHO
SANS
DWAF



Note: A and B shows pH, EC, TDS, DO, Alkalinity, and TH for groundwater and surface water. C and D shows mean concentrations of major ions for groundwater. E and F shows mean trace element concentrations for groundwater and surface water.

***Figure 4.1** Bar charts depicting mean concentrations of parameters for groundwater, surface water and leachate over wet and dry seasons plotted with WHO, SANS, and DWAF guidelines. Leachate shows high concentrations than groundwater and surface water from charts A-F. High concentrations signal contamination with all measured parameters in comparison to national and international standards.*

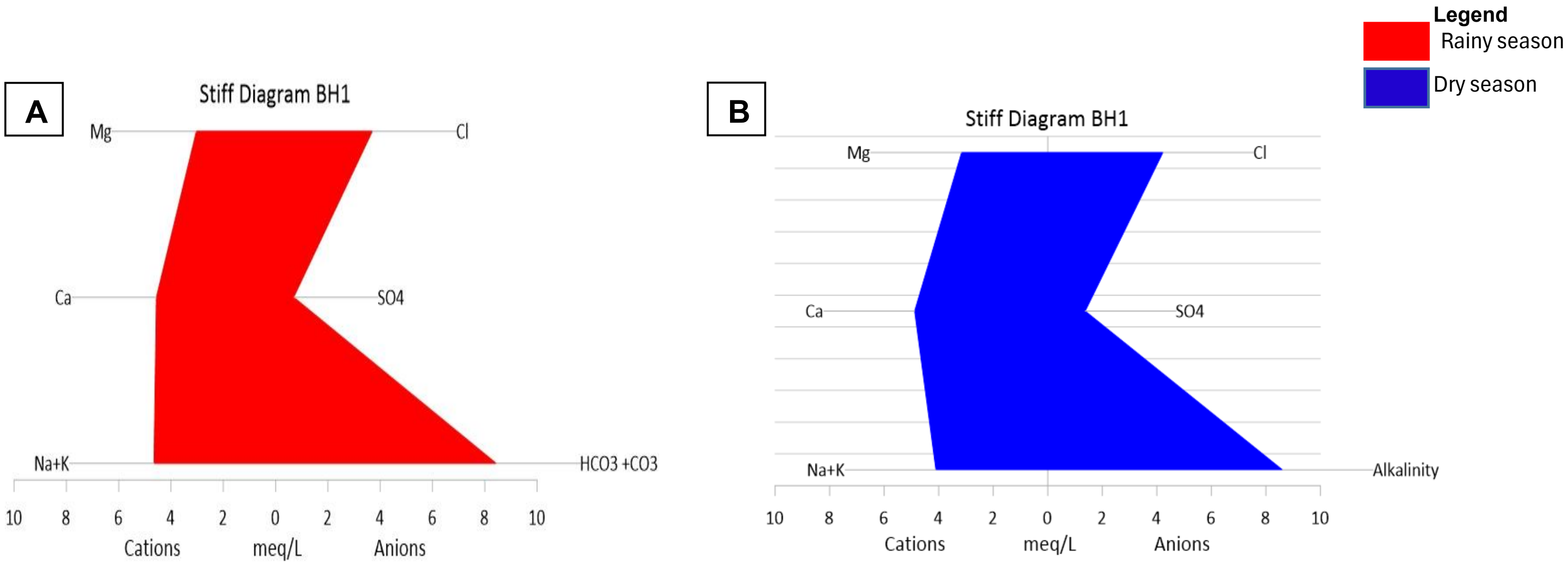
4.3.2 Electrical conductivity and total dissolved solids

Groundwater had a mean EC concentration of 71,20 mg/ℓ during the wet season and 69,20 mg/ℓ during the dry season. The mean values in surface water were 30,30 mg/ℓ and 30,70 mg/ℓ, respectively. EC concentrations were below the WHO and SANS drinking water requirements for both groundwater and surface water throughout the year, except during the dry season, when only groundwater exceeded the DWAF limits, rendering the water unfit for agriculture (Figure 4.1a-b). Both groundwater and surface water had TDS levels that were within the WHO and SANS guidelines as depicted by Figure 4.1a-b. The mean values for groundwater in the wet and dry seasons were 375,00 mg/ℓ and 393,50 mg/ℓ, respectively. In contrast, surface water had mean values of 151,10 mg/ℓ and 153,00 mg/ℓ in both seasons. These results for EC and TDS in groundwater were lower than those reported in Bloemfontein, South Africa (Makhadi et al., 2020) and Buffalo City, South Africa (Nyika et al., 2022). However, the mean EC content was significantly greater than that reported in groundwater near Varanasi, India (Mishra et al., 2019). Other investigations have revealed low TDS and EC levels in surface water (Amano et al., 2020). In this case, the value of EC (Table 4.2) in these water sources near the landfill site could imply a reduced influence from leachate. Makhadi et al. (2020) concluded that high levels of EC are caused by high concentrations of ionic components in the water. Low EC in water bodies near the Botshabelo waste site may indicate a low concentration of ionic components. TDS is regarded as a composite indication of both a wide variety of chemical contaminants and the aesthetic quality of drinking water. It includes both inorganic and organic particles in water (Palomeras et al., 2020). Given that all of the water sources had average TDS values below permissible limits, it may be concluded that there was no major pollution from runoff from a nearby garbage site. Ololade et al. (2019) found that high temperatures promote weathering, ion-exchange capacity, dissolution, and desorption. As a result, it is conceivable that little rainfall supported limited breakdown, resulting in low TDS in groundwater (Makhadi et al., 2020).

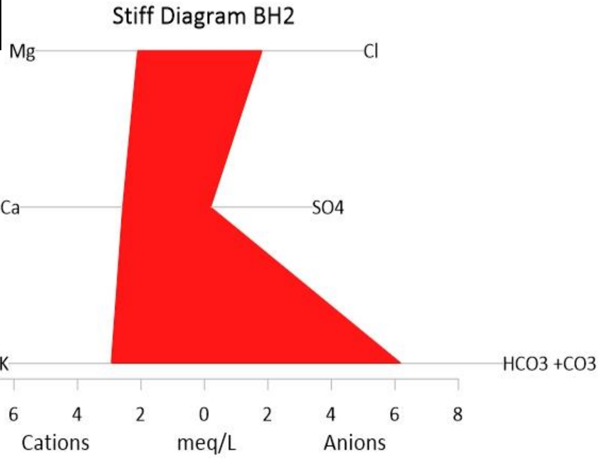
4.3.3 Alkalinity

The alkalinity of water is its ability to neutralise acids and bases that are introduced to it, acting as a buffer and allowing it to maintain sufficient pH levels. Alkalinity is determined by the amount of calcium carbonate in the water, which has a neutralising impact (Naveen et al., 2016). The mean alkalinity concentrations in groundwater throughout the wet and dry seasons were 295,17 mg/ℓ and 325,69 mg/ℓ. Amano et al. (2021) found lower amounts of alkalinity in their investigation of groundwater quality around a landfill, whereas Rashid et al. (2022) found higher mean values of more than 400 mg/ℓ for two wells. According to Reddy and Nandini (2011), high levels of alkalinity give the water a disagreeable flavour, rendering it unfit for drinking. The mean value of 295,90 mg/ℓ indicates that the water can sustain acceptable pH levels, making it suitable for use. There are no alkalinity requirements provided by WHO, SANS, or DWAF in Table 4.2. The landfill site had no pollution effect on groundwater because groundwater concentrations were relatively low in comparison to leachate, despite less substantial differences in concentrations between wet and dry seasons (Figure 4.1a). This can be linked to the geology of the area, which is characterised by mudstone and shale, as well as the clay cover, which functions as a pollution trap and a protective buffer against contaminants (Nyika et al., 2019).

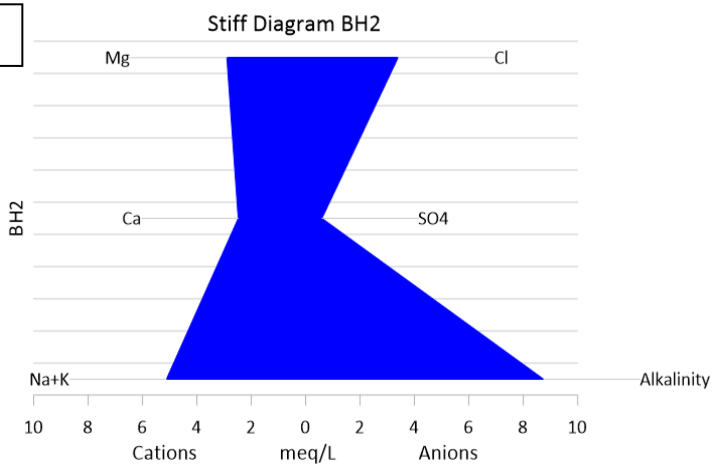
Surface water at the landfill site had mean values of 167,69 mg/ℓ and 167,52 mg/ℓ in both the wet and dry seasons, which were significantly lower than groundwater (Figure 4.1b). These concentrations were significantly lower than those reported by Gwisai et al. (2019) at 600 mg/ℓ in the Gaborone landfill in Botswana. Amano et al. (2021) found slightly lower quantities of alkalinity in their investigation. Similarly, there were no substantial changes in surface water concentrations due to seasonal variation, and no leachate pollution from runoff was detected. According to Raju et al. (2014), reduced alkalinity in surface water is caused by the action of microorganisms in water that emit carbon dioxide and organic matter, which reduces pH and alkalinity. Moreover, surface water is more exposed to the environment than groundwater. The Stiff diagrams depict higher alkalinity Bicarbonate + Carbonate (HCO_3 + CO_3) concentrations in all groundwater samples, suggesting an alkaline water as opposed to lower values in surface water (Figure 4.2a-l).


A
Stiff Diagram BH1
Mg
Cl
Ca
SO4
Na+K
HCO3 +CO3
10 8 6 4 2 0 2 4 6 8 10
Cations
meq/L
Anions
B
Stiff Diagram BH1
Mg
Cl
Ca
SO4
Na+K
Alkalinity
10 8 6 4 2 0 2 4 6 8 10
Cations
meq/L
Anions
Legend
Rainy season
Dry season

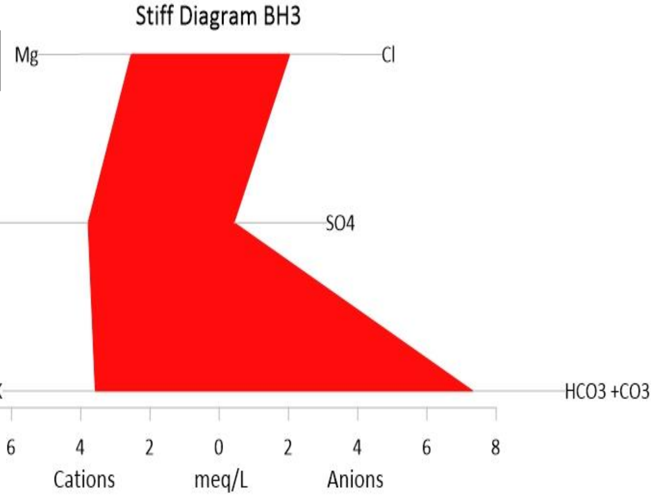
C


Stiff Diagram BH2
Mg
Cl
Ca
SO4
Na+K
HCO3 +CO3
8
6
4
2
0
2
4
6
8
Cations
meq/L
Anions

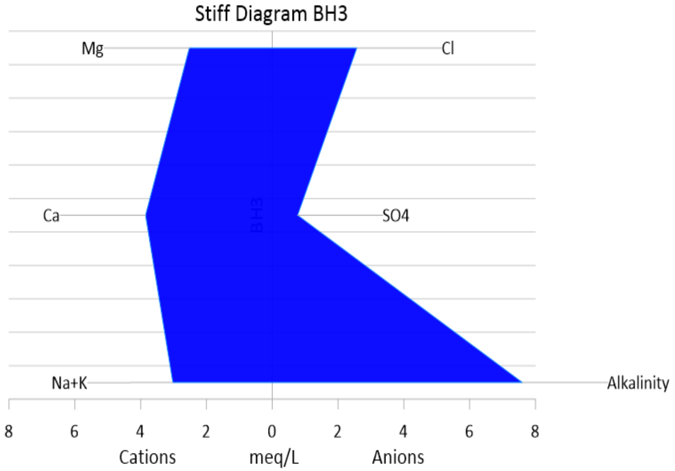
D

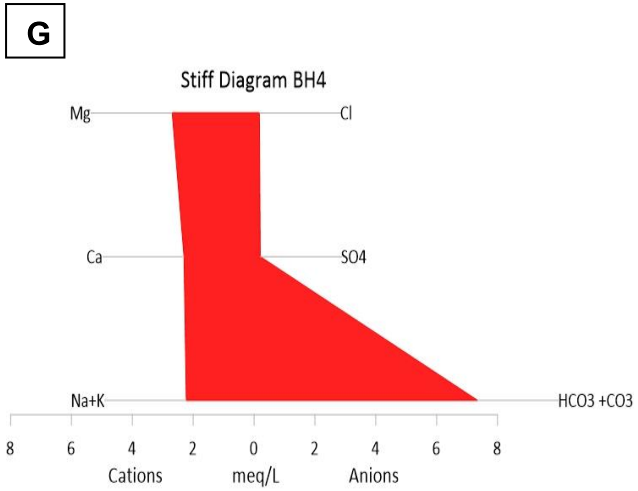

Stiff Diagram BH2
BH2
Mg
Cl
Ca
SO4
Na+K
Alkalinity
10
8
6
4
2
0
2
4
6
8
10
Cations
meq/L
Anions

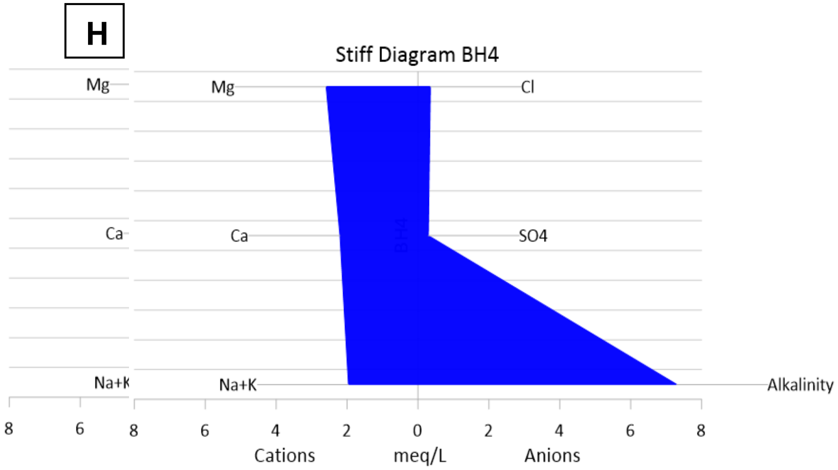
E

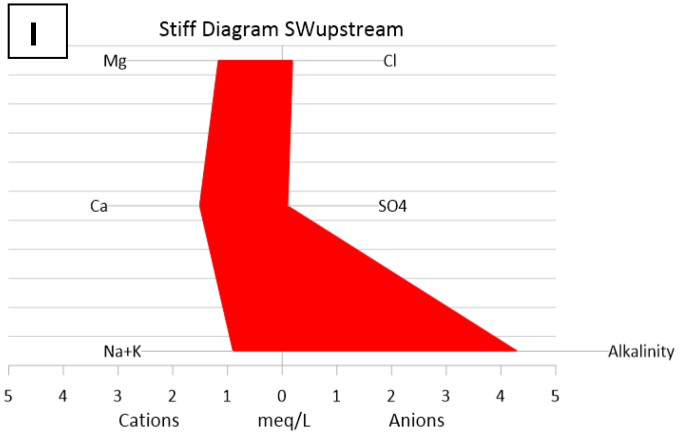

Stiff Diagram BH3
Mg
Cl
Ca
SO4
Na+K
HCO3 +CO3
8
6
4
2
0
2
4
6
8
Cations
meq/L
Anions

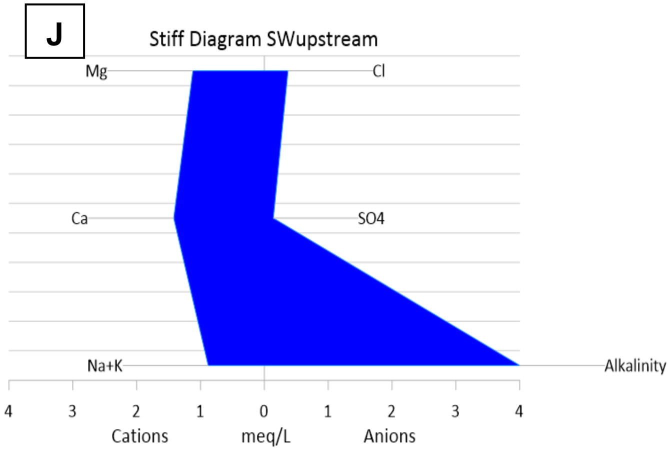
F

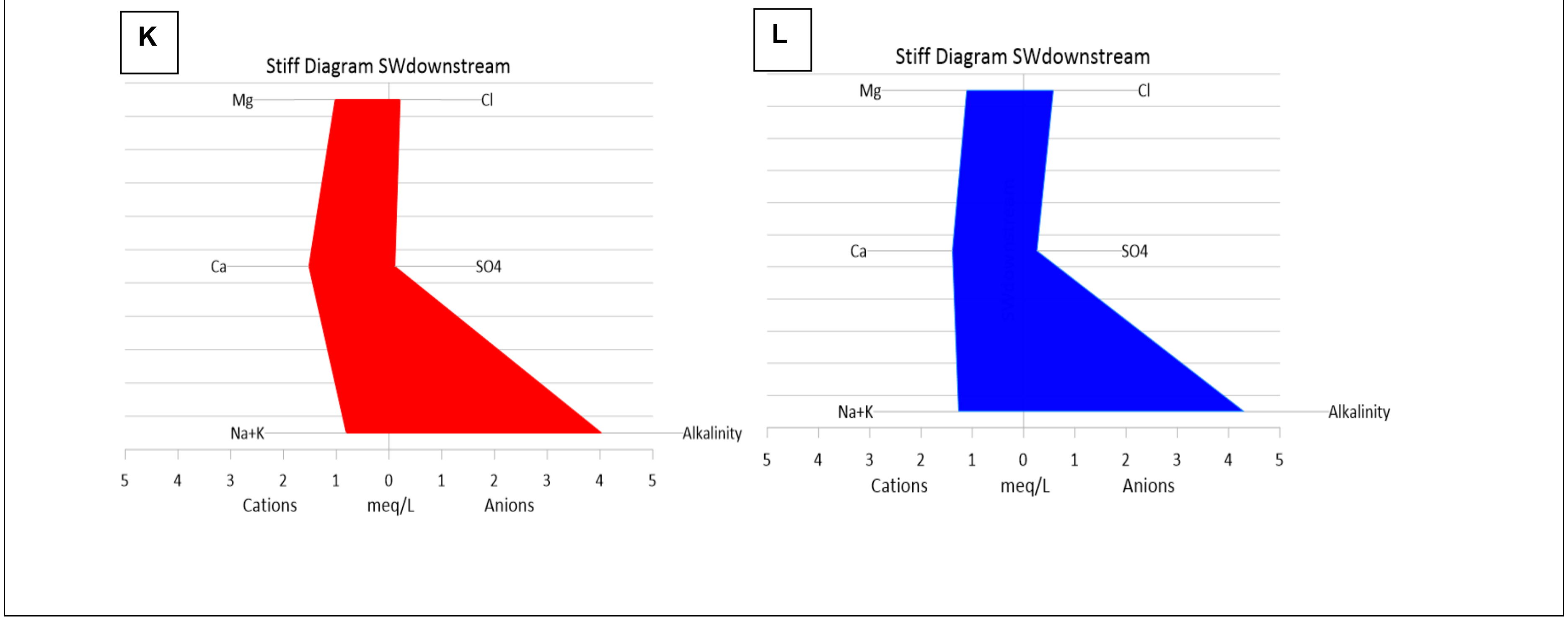

Stiff Diagram BH3
Mg
Cl
Ca
SO4
Na+K
Alkalinity
8
6
4
2
0
2
4
6
8
Cations
meq/L
Anions


G
Stiff Diagram BH4
Mg
Cl
Ca
SO4
Na+K
HCO3 +CO3
8
6
4
2
0
2
4
6
8
Cations
meq/L
Anions


H
Stiff Diagram BH4
Mg
Mg
Cl
Ca
Ca
SO4
Na+K
Na+K
Alkalinity
8
6
8
6
4
2
0
2
4
6
8
Cations
meq/L
Anions


I
Stiff Diagram SWupstream
Mg
Cl
Ca
SO4
Na+K
Alkalinity
5
4
3
2
1
0
1
2
3
4
5
Cations
meq/L
Anions


J
Stiff Diagram SWupstream
Mg
Cl
Ca
SO4
Na+K
Alkalinity
4
3
2
1
0
1
2
3
4
Cations
meq/L
Anions



Note: A−L depicts the major ions and alkalinity as well as changes in concentrations shown by varying shapes measured in Milliequivalent per litre (meq/ℓ).

Figure 4.1 Stiff diagrams for groundwater and surface water in wet and dry seasons *(Designed by author using Grapher Golden Software)*

4.3.4 Total hardness

Singh et al. (2022) defined TH in water as the concentration of Ca and Mg ions present. Water hardness is related to its household use for cleaning with soap since it must accept the soap concentration. SANS, WHO, and DWAF have not stated any criteria for drinking and irrigation in this study. The study's results revealed mean values of 295,17 mg/ℓ for the wet season and 307,51 mg/ℓ for the dry season, respectively (Table 4.2). In their study, Ololade et al. (2019) observed that borehole samples close to the landfill site at 50 m had high total alkalinity concentrations, thus supporting this study. Groundwater around the Oti-Dompoase dump in Ghana was also found to have TH exceeding 200,00 mg/ℓ, which corresponds to the findings of this study, indicating hard water, which is similar to the study of Amano et al. (2020) and lower than the one reported in Varanashi in India (Mishra et al., 2019). The results of this study demonstrate that the groundwater is extremely hard. The existence of very hard water in the area can be linked to the geology, which is characterised by sedimentary rocks such as siltstones interbedded with layers of clay, sandstone, and shale (Makhadi et al., 2020). As a result, the percolation and flow of water through these rocks may lead to the leaching of divalent ions, such as Ca_2+ and Mg_2+, in the groundwater, suggesting differing source origin from the leachate itself (Figure 4.2a).

In contrast, surface water exhibited mean values of 127,28 mg/ℓ and 125,63 mg/ℓ for both the rainy and dry seasons (Table 4.2). TH also did not change drastically over the two seasons, which implies no seasonal impact on its concentration. The exposure of surface water to the environment where dilution of ions by rainfall takes place, coupled with the susceptibility of surface water to chemical and biological reactions, is the reason for its lower concentrations. Sengupta (2013) categorised water into four classes based on hardness: soft at <60; moderately soft at 61−120; hard at 121−180; and very hard at >180. Hardness concentrations greater than 200 mg/ℓ can cause scale deposits in tanks, pipes, and treatment plants, as well as calcium carbonate scale (WHO, 2011). According to the WHO, severe hardness, defined as dissolved solids greater than 500 mg/ℓ, might impair the taste of water. As a result, overall hardness levels less than 500 mg/ℓ for all water samples at the landfill site indicate a relatively fair water taste.

4.3.5 Dissolved oxygen

DO is the amount of oxygen available to all aquatic organisms. The mean value of DO in groundwater was 4,38 mg/ℓ and 4,60 mg/ℓ over two seasons, while surface water was 3,44 mg/ℓ and 3,46 mg/ℓ, with 5 mg/ℓ (Table 4.2), being comparable to a study conducted in Revdalen, Norway, which reported low DO in groundwater near a landfill site (Abiriga et al., 2020). In another investigation, Alam et al. (2021) found low DO concentrations of 5 mg/ℓ in surface water samples from a landfill in Bangladesh. Low DO levels may be detrimental to aquatic species (Khan and Wen, 2021), and animals such as fish cannot survive for long in water with DO levels below 5 mg/ℓ (Shoeb et al., 2022). When oxygen levels are low, nutrients bound to bottom sediments can be released into the water column, allowing more plankton growth and, eventually, increased oxygen depletion. Furthermore, a low quantity of DO in the water suggests that a nearby landfill may have contaminated the water by releasing a large amount of organic debris and causing an oxygen shortage.

4.3.6 Cations

- **Calcium and magnesium**

Both Ca and Mg are found in water, but Ca has higher natural amounts. According to Erdogan et al. (2019), Ca and Mg in groundwater originate from mafic rocks, including minerals such as amphiboles, feldspars, pyroxenes, and garnet. Ca is an alkaline earth metal that occurs in varying amounts in water, along with Mg to form water hardness (DWAF, 1996). The results of this investigation showed that all groundwater had mean values of 66,26 mg/ℓ and 67,09 mg/ℓ in both wet and dry seasons (Table 4.2). The groundwater had concentrations below the WHO standards, rendering them fit for drinking purposes (Figure 4.1c). Rinae et al. (2018) discovered Ca concentrations that were elevated beyond the guidelines set for drinking purposes, thus, showing contamination by the landfill leachate. However, this study revealed leachate concentration to be far below the borehole samples mentioned; hence, the detected values of these ions can be attributed to another complex source or the geology of the area where dolerite dykes, being mafic in nature, contain Ca-bearing minerals like feldspars that could have interacted with groundwater during intrusions to bring about Ca-enrichment. In their investigation, Nagarajan et al. (2012) determined that the origin of ions in groundwater (including Ca) was due to geochemistry rather than leachate at the landfill. Surface water samples showed mean values of 30,20 mg/ℓ and 28,25 mg/ℓ for both wet and dry seasons, indicating that all surface water samples were below the acceptable limits set by WHO for drinking water. There are no specifications made by SANS for drinking water and DWAF for irrigation qualities. In both seasons, the landfill had no pollution effect on the surface water. Lasagna et al. (2020) asserted that the differences in temperature and rainfall can alter the quality of surface and groundwater in that runoff by high rainfall can transport pollutants like heavy metals, pesticides, and phosphorus into lakes and streams, thus deteriorating the water quality. For this study, the average yearly rainfall in Botshabelo is about 533 mm per year, which is far below the world average equalling 860 mm per year, and it records the highest precipitation of about 323 mm monthly in summer as well as 83 mm in winter; thus, indicating that it is a semi-arid area (Dennis and Dennis, 2011). Consequently, this can suggest low leachate generation rates that do not run off to reach the surface water stream near the landfill site. The Stiff plots also confirmed higher Ca values in groundwater samples as opposed to surface water samples similar to observations made for previously mentioned parameters (Figure 4.2c-d).

Mg concentrations in groundwater averaged 31,50 mg/ℓ and 34,00 mg/ℓ during the wet and dry seasons, respectively. All the groundwater around the landfill was slightly above the WHO drinking quality threshold for both rainy and dry seasons. No standard limits were allocated for Mg by SANS for drinking quality and DWAF for irrigation quality. The Stiff plots (Figure 4.2a-h) showed little to no changes in Mg concentrations. Nagajaran et al. (2012) reported Mg contamination in groundwater in India where average concentration for Mg were 55,0 mg/ℓ, much higher when compared to this study. Slightly higher Mg concentrations were recorded by Kanmani and Gandhimathi (2013). As stated above, the contamination of Mg in groundwater can be geological in origin derived from mafic rocks that contain garnet, amphiboles, pyroxenes and feldspars (Erdogan et al., 2019). The elevated concentration of cation, or Mg_2+, in groundwater may be linked to the local geology, which consists of sedimentary rocks with a series of dolerite intrusion.

The dissolution and degradation of inorganic solid wastes such as tiles, plaster and concrete from the landfill sites may lead to the higher concentrations of cations in surrounding water bodies (Nyika et al., 2022). Moreover, the leachate was also contaminated with Mg, which in turn can pollute groundwater due to the geohydrology of the study area. Based on the maps of the Department of Water and Sanitation

(DWS), the research region consists of fractured or intergranular aquifers, which can facilitate pollutant transportation into groundwater. Gemail et al. (2022) studied the groundwater quality at a landfill near a highly fractured aquifer basalt, where they observed high contamination of groundwater by the leachate as a result of flow paths formed by fractures. Additionally, their borehole depths ranged from 7 m to 15,3 m in depth, which they classify as a shallow aquifer depth. Similar to the findings of Gemail et al. (2022), the depth to groundwater in the study area ranged from 3,1 m to 10,5 m, indicating that it is a shallow aquifer that can encourage pollution of groundwater due to contaminants' short travel distances. In terms of irrigation, Guo et al. (2015) stated that Mg is an important nutrient for plant growth and development when provided at optimal amounts in the soil. However, excessive levels might cause plants to develop shorter roots and shoots, inhibiting growth.

- **Sodium**

The presence of Na in the environment is quite extensive and it occurs in various forms such as a solid in the form of sodium Cl, sodium sulphate, and HCO_3 (DWAF, 1996). It is the earth's sixth most prevalent element found in plants, soils, water, and food (USEPA, 2003). It occurs in the majority of mineral deposits in the environment, and due to its solubility, it is found in larger concentrations in groundwater than in surface water. The findings of this investigation revealed mean Na concentrations of 75,11 mg/ℓ during the wet season and 79,09 mg/ℓ during the dry season. For drinking quality, the groundwater fell below the threshold given by WHO for both seasons, thus meeting the standards for drinking water purposes for Na, but exceeding the DWAF limits (Figure 4.1c-d). A study by Palomeras et al. (2021) indicated Na concentrations of roughly 174,15 mg/ℓ for both rainy and dry seasons, which was substantially higher than in this study. Their research revealed concentrations that fell below the WHO limits; hence, deeming groundwater suitable for drinking without risk to humans by consumption. Nagarajan et al. (2012) found contrasting results, reporting salt levels that surpassed the WHO standards at 437 mg/ℓ due to their closeness to the landfill. Excess Na concentrations are harmful to the physical qualities of soil and plants, resulting in toxicity and limiting plant growth (DWAF, 1996). Furthermore, high concentrations of Na cause soil dispersion and swelling, resulting in poor soil drainage and increased surface runoff (Ololade et al., 2019).

According to Khan et al. (2020), excess Na is a result of anthropogenic activities rather than weathering of rocks and minerals. High concentrations of Na can signal pollution and may require treatment before use. Moreover, the presence of Na salt in anthropogenic waste dumping may explain the elevated Na level in comparison to the DWAF (1996) requirements. Major cations and anions are within the allowed limit, indicating low pollution problems and minimal impact on water chemistry at the Botshabelo non-engineered landfill site. Surface water Na concentrations ranged from 17,92 mg/ℓ to 21,28 mg/ℓ in both the wet and dry seasons. All surface water samples were below the WHO and DWAF criteria, thereby making the water acceptable for consumption and irrigation (Figure 4.1c-d). The Stiff diagrams in Figure 4.2 show supporting low concentrations of Na compared to other ions for all water samples. Figure 4.3 depicts the water sample findings on the United States salinity diagram for irrigation. Boreholes BH2 and BH3 were classified as C2−S2 for medium salinity and Na risks. BH1 dropped in C3−S3, indicating high salinity and Na hazards, while BH4 fell in C3−S2, indicating high salinity and mild Na hazards. Similar examples of elevated salinity and salt dangers were observed in groundwater near a dump site in Bloemfontein (Ololade et al., 2019). High-Na water can yield harmful amounts of exchangeable Na. These conditions may eventually harm the soil structure, requiring particular soil management (Saha et al., 2019). Furthermore, medium Na water in BH4 provides a clear Na hazard in fine-textured soils with high cation-exchange capacity, particularly in the absence of gypsum, but can be employed in coarse-grained or organic soils with poor permeability (Dhok et al., 2011). All surface water samples plotted in the C2−S1 field showed medium salinity and low Na risks, making them appropriate for irrigation.

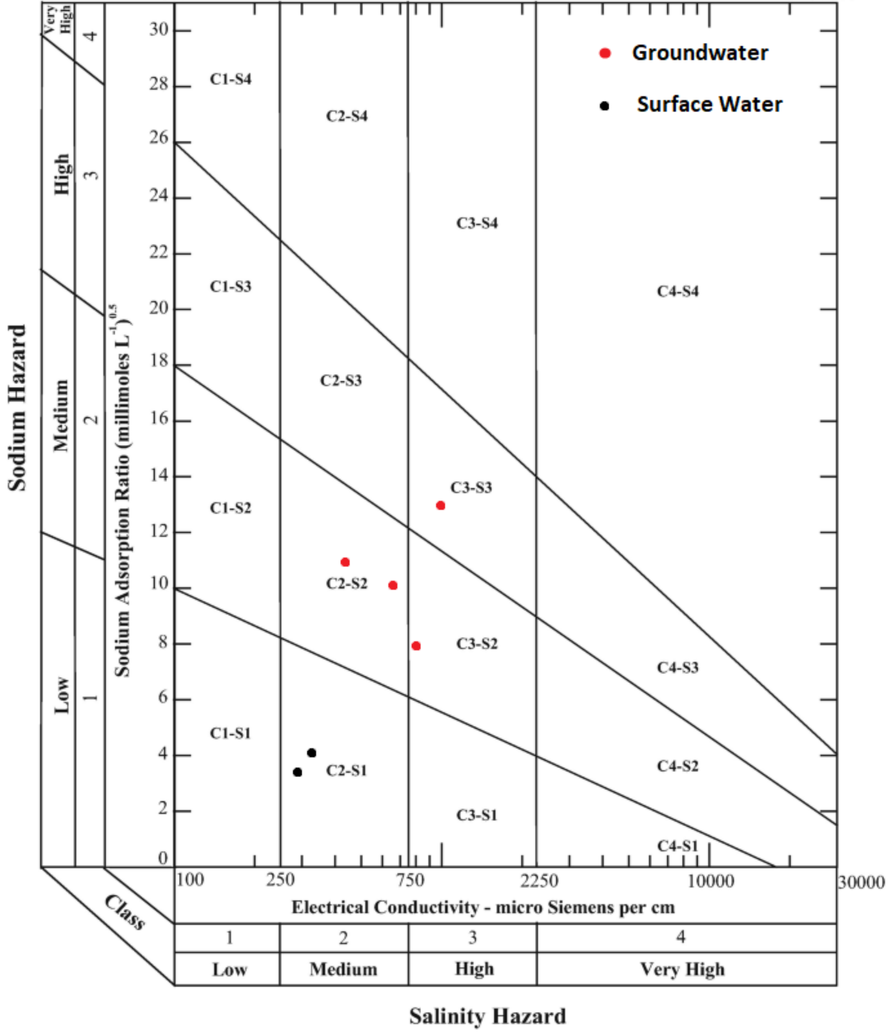


Figure 4.3 Salinity hazard diagram for groundwater and surface water in both seasons
(Plotted by author using Wilcox plot template; Adopted from Elmanshawy et al., 2024)

K showed very low concentrations in all groundwater and surface water over two seasons. The standards for both WHO and SANS for drinking water quality for K are 300 mg/ℓ and 200 mg/ℓ, while DWAF showed no limit for irrigational use for K. Groundwater K concentrations averaged 3,27 mg/ℓ and 4,91 mg/ℓ during the wet and dry seasons, respectively. Rashid et al. (2022) measured extremely high mean concentrations of 1 756 mg/ℓ over four seasons at an open dump in Kurdistan, Iraq. Meanwhile, Ololade et al. (2019) found somewhat higher K amounts than this study in Bloemfontein. In contrast, surface water had mean concentrations of 3,12 mg/ℓ and 5,81 mg/ℓ during the rainy and dry seasons, respectively. Both groundwater and surface water were deemed safe for drinking. The Stiff diagrams reveal low K contents for all of the water samples. Though the leachate concentration was significantly higher than that of all water samples, no pollution occurred, and the clay layer in the landfill site, like alkalinity, could have acted as a pollutant trap to protect groundwater.

Furthermore, the Botshabelo area's minimal rainfall causes low flow, preventing leachate from reaching surface water. According to WHO (2009), K is widely distributed in the environment and poses no risk to humans when consumed in accordance with the guidelines stated. Furthermore, the WHO (2009) confirmed that the concentration of K in clean groundwater is scarcely high. Excess K exerts similar effects on the structure of irrigated soil, causing it to become more compact and denser, preventing water infiltration (Xu et al., 2020). As a result, water does not reach the roots of plants, hindering growth. Several studies have also found very low to undetectable quantities of K at landfill sites (Makhadi et al., 2018; Reddy and Nandini, 2011; Sivakumar et al., 2015).

4.3.7 Anions

- **Chloride**

Cl is very common in water, and being soluble, it tends to accumulate. Its concentrations in fresh water reach a few hundred of mg/ℓ (DWAF, 1996). Typically, Cl is found in irrigated water as return flows and industrial and sewage effluents and does not occur in nature. Additionally, it is one of the determinants of groundwater contamination by the leachate (Kamble et al., 2020). The current investigation found mean concentrations of 68,53 mg/ℓ and 93,25 mg/ℓ throughout the rainy and dry seasons, respectively. Although the leachate had high levels of Cl, it had no environmental impact on groundwater around the landfill site. The groundwater in the dump fell within the WHO and SANS requirements for drinking quality for both seasons, pronouncing them suitable for drinking purposes, while exceeding the DWAF limitations for irrigation purposes (Figure 4.1c-d). The Stiff diagrams were associated with lower Cl values in all groundwater samples (Figure 4.2a–l). El-Salam and Abu-Zuid (2015) investigated the influence of landfills on groundwater. They discovered Cl values ranging from 2 240 mg/ℓ to 11 750 mg/ℓ, which was due to the incursion of a very salty leachate. Amano et al. (2021) obtained results similar to those of the current investigation. Furthermore, the study's findings revealed that Cl concentrations declined as distance from the dump increased, demonstrating that landfill distance/proximity influences Cl. Amano et al. (2021) obtained contradictory results, revealing increasing Cl concentrations as distance from the dump increased. Excessive Cl concentrations may cause leaf burn in crops because it is a soluble salt that is not adsorbed by the soil; therefore, it is taken up by plants through the roots to the leaves (DWAF, 1996). Additionally, Cl is a crucial plant nutrient that does not generally cause harm to plants as opposed to other nutrients.

Surface water, on the other hand, had mean Cl concentrations of 7,10 mg/ℓ and 17,05 mg/ℓ throughout the wet and dry seasons. In both seasons, all surface samples fell below the WHO, SANS, and DWAF drinking and irrigation water quality standards (Figure 4.1c-d). Reggam et al. (2017) found that the average Cl concentration in surface water samples was 392,49 mg/ℓ, and they concluded human activity from industrial and urban activities. Additionally, Khan et al. (2020) reasoned that high concentrations of Cl (normally greater than 1 000 mg/ℓ), can be attributed to leachate contamination and high temperatures with very less rainfall. However, this study proved to differ from these findings due to very low concentrations of Cl in both groundwater and surface water, which indicated no pollution impact by the leachate. Seasonal influence may have played a substantial role in the overall amounts of Cl in both groundwater and surface water, as concentrations are significantly higher in the dry season than in the rainy season. According to Vongdala et al. (2019), the dry season can result in the formation of a concentrated leachate due to reduced rainfall and evaporation. Also, both groundwater and surface water remain connected with each other; thus, during dry seasons groundwater can recharge surface water, resulting in elevated Cl concentrations.

- **Nitrate and nitrite**

NO_3^- occurs in the natural environment and it is also a vital nutrient for the growth of plants in which it is found in different concentrations (WHO, 2017). NO_2^- is also found in the environment and is a product of reduction of NO_3^- by the depletion of oxygen by microbial activity (WHO, 2017). In terms of its stability, NO_3^- is more stable in an oxidation state; hence, has concentrations higher than those of NO_2^-. The study's findings revealed that mean NO_3^- and NO_2^- concentrations in the rainy season were 0,16 mg/ℓ and 0,01 mg/ℓ, respectively, whereas in the dry season they were 0,39 mg/ℓ and 0,01 mg/ℓ. Kamble et al. (2020) found much higher NO_3^- concentrations in groundwater samples from the Jawaharnagar dumpsite in India during both wet and dry seasons than in this investigation. However, Nguyen and Huynh (2023) discovered quantities of NO_3^- and NO_2^- that were below detection limits in groundwater, which were similar to the current investigation.

All samples were in the permissible limits for drinking quality and showed no NO_3^- pollution (Figure 4.1c). DWAF does not have a specific guideline value for irrigation water quality, but according to Han et al. (2014), NO_3^- concentrations at the values of 42 mg/ℓ had polluted groundwater in an extensive manner due to agricultural lands close to the Zhoukou landfill site in China. Surface water likewise had very low quantities of NO_3^- and NO_2^-, with averages of 0,16 mg/ℓ and 0,01 mg/ℓ during the rainy season, and 0,16 mg/ℓ and 0,01 mg/ℓ during the dry season. Alam et al. (2021) investigated the surface water quality near a dump site and found a mean value of 30 mg/ℓ of nitrate. Ololade et al. (2019) reported NO_3^- and NO_2^- values that were within the acceptable limits of SANS 241 for drinking water quality. Similar values of NO_2^- compared to this study were made by Seiyaboh and Izah (2017), who conducted a study to review the effect of anthropogenic activities on surface water. They concluded that for fresh water, the usual values of NO_2^- were less than the permissible limits set (Figure 4.1d). The Botshabelo landfill site is located on the agricultural area that did not show any pollution by fertilisers due to relatively low NO_3^- values that are typical of unpolluted waters.

Groundwater and surface water were both below the acceptable limits established by WHO and SANS for drinking purposes; however, there are no limits for irrigation by DWAF.

- **Sulphate**

SO_4 occurs extensively in natural waters in variable quantities that range in 10s to 1 000 mg/ℓ, whereby groundwater typically has the highest concentrations because of different sources that are geochemical and biological in nature (WHO, 2017). Moreover, about 30% of SO_4 originates from the atmosphere, while the remainder is due to biological and geological processes. From the results of this study, the same is true as groundwater contains higher concentrations of SO_4 than surface water (Figure 4.1c-d). The groundwater around the dump site had mean SO_4 concentrations of 36,51 mg/ℓ and 36,63 mg/ℓ in the wet and dry seasons, respectively. According to WHO (2017) and SANS (2015), the highest SO_4 threshold is 500 mg/ℓ; thus, all of the samples were within the acceptable drinking limits. Rashid et al. (2022), Kanmani and Gandhimathi (2013), and Kamble et al. (2019) found higher SO_4 concentrations ranging from 45 mg/ℓ to 250 mg/ℓ in different seasons, which were acceptable according to WHO criteria. The study's findings in both seasons revealed that SO_4

caused no pollution due to the low quantities found in the leachate, which were even lower than those in groundwater.

Surface water samples had 100% very low amounts of SO_4, indicating no risk of pollution. The surface water had mean amounts of SO_4 that were lower than the standards specified for drinking water quality, suggesting that the water could be safe to drink. Surface water concentrations averaged 4,97 mg/ℓ and 10,15 mg/ℓ during the rainy and dry seasons, respectively. These low concentrations depicted no impact of the municipal landfill on surface water. Igbanoi et al. (2019) reported SO_4 concentrations that were much higher than the present study but posed no pollution at a landfill in the Niger Delta in Ghana. They recorded mean values of 85 mg/ℓ. High concentrations of SO_4 are associated with mining areas where acid mine drainage takes place due to oxidation of pyrite (Meride and Ayenew, 2016). Also, elevated phosphate concentrations in water, particularly surface water, can cause phosphorus to be released into stream water, promoting algal growth (Ayana, 2019). SO_4 is a vital nutrient for plant growth when it has suitable concentrations in irrigation water (Bauder et al., 2014), so groundwater and surface water can be used for irrigation.

4.3.8 Trace elements

Heavy metals and trace metals are metallic in nature and have specific gravity of ≥5 cm^3. They can cause pollution to groundwater and surface water even at trace amounts when they come into contact with them (Izah et al., 2016). Because of their stability, heavy metals cannot undergo chemical or biological degradation; instead they are mobilised by surface runoff, discharge of waste into surface waters, soil erosion as well as other anthropogenic sources (Izah et al., 2016). Also, municipal landfill sites are areas where toxic heavy and trace elements have been discovered to form widely due to the leachate that comes into contact with the waste that picks up these toxins to land them in the environment (Essien et al., 2022). Although leachate generation can reduce in the landfill, trace and heavy metals can still pose serious effects to the environment due their pervasive accumulation in surface and groundwater. Al, Sb, Ba, As, B, Se, Cd, Co, Cr, Cu, Fe, Ni, Pb, V, Zb, Mo, and F were among the trace metals found in groundwater and surface water samples throughout both seasons. However, Mn and U recorded concentrations that exceeded the water quality standards for drinking by the WHO and DWAF, but below SANS (Figure 4.1e-f). The mean concentrations for Mn over both seasons were 0,24 mg/ℓ and 0,22 mg/ℓ, resulting in contamination of groundwater around the landfill site. In a study by Han et al. (2013) in a landfill in India, groundwater recorded excessive Mn concentrations (>0,20 mg/ℓ) that exceeded WHO limits. Similarly, Abiriga et al. (2020) found mean values of 4,4 mg/ℓ as a result of landfill pollution on groundwater. Kanmani and Gandhimathi (2013) observed high trace element enrichment due to leachate migration near the waste column; similary, groundwater may have been affected by leachate migration over time, which makes it unsuitable for drinking and irrigation. Mn can be derived from human activities such as landfill, industrial discharges and mining activities, which land in water sources (WHO, 2020). Sources of Mn in a landfill site include pigments, bottle caps, paints, blades, and insecticides (Beinabaj et al., 2023). According to Usman et al. (2021), Mn occurrence is associated with element substitution of iron-bearing minerals like manganite or rhodochrosite. Excess Mn in water can create a colour change in which it loses clarity and becomes dark. Surface water around the dump site had mean amounts of 0,08 mg/ℓ and 0,01 mg/ℓ in the wet and dry seasons, respectively. There was, however, no pollution posed by Mn on surface water as it fell below the national and international standards over all seasons (Figure 4.3e-f). Ololade et al. (2019) also reported no Mn enrichment for both autumn and winter in surface water samples, but contradicts with the results of this study for wet season.

U occurs widely distributed in nature in granitic rocks and minerals. It is actively used as fuel for electricity at power stations, and is introduced into the environment due to leaching, emissions, fertilisers, coal combustion, and from tailings (WHO, 2022). In this study, also, U slightly polluted groundwater samples. The study found mean U concentrations of 0,06 mg/ℓ in the wet season and 0,01 mg/ℓ in the dry season. The findings demonstrated groundwater contamination only during the wet season, with mean concentrations above the WHO and DWAF standards but not the SANS limit (Figure 4.1e-f). U enrichment has been reported by Zupunski et al. (2023) in groundwater from gold tailings in nearby communities in Gauteng. All surface water samples had zero U detection, signalling no U pollution for both seasons. According to the WHO (2022), crop production decreases by 50% when irrigated water contains 50 mg/ℓ U, and at 10 mg/ℓ, no crop yield effect has been recorded. In contrast, all groundwater and surface water samples in the dry season had trace amounts that were below all the water quality standards, indicating no contamination at all. These waters can be used for both drinking and irrigation. In addition, U in leachate was in trace amounts, suggesting that no U-containing wastes were deposited in the landfill site and that the landfill posed no effect on water quality regarding U. Therefore, the enrichment of U at the landfill site may be more geogenic than anthropogenic. According to Letman et al. (2018), U occurs in rock formations that often cause its enrichment in groundwater in specific geochemical conditions. For instance, the movement of U into groundwater has been linked to processess such as ion-exchange, weathering of U-rich rocks, generation of soluble complexes between constiuents present in groundwater and uranyl ions, as well as changes in redox conditions and pH. Other sources, though, may be anthropogenic. For instance, Sjöblom et al. (2013) studied the U pollution at a landfill site that was from a U mine as well as ammunition facilities in Russia. The most significant finding in this study was that all U trace elements had larger concentrations during the wet season than during the dry season, when they were relatively low. In a study conducted by Zloch et al. (2018) at the Zdounky-Kuchynky landfill site in the Czech Republic, they detected a rise in the trace element in the leachate during the rainy season in February and a decrease in concentrations in a dry season in June. They then concluded that seasonal changes resulted in leachate generation in a rainy season that could have taken up trace elements from the waste and infiltrated down, along with the leachate into groundwater. A similar phenomenon may have also taken place for this study where there were trace element enrichment in groundwater during the wet, rainy season in the past, whereas there was no enrichment due to less to no rainfall during the dry season.

4.4 Hydrochemistry

Piper's (1944) diagram depicts the geochemical history of water. In Figure 4.4, all water samples are displayed in the trilinear diagram in various fields for both the rainy and dry seasons. Hydrochemistry helps to determine the type of water by revealing the dominant ions that the water contains in order to track its composition, properties, and behaviour (Gautam and Rai, 2023). Therefore, Piper plots were constructed to illustrate the hydrogeochemical facies in groundwater and surface water around the Botshabelo non-engineered landfill site. The results showed that groundwater and surface water samples around the Botshabelo waste site lie in quadrants 1 (BH1, BH2, and BH3) and 4 (BH4, SWu and SWd) of the Piper diagram. Quadrant 1 representing BH1, BH2 and BH3 samples, indicates Calcium Magnesium Bicarbonate ($CaMgHCO_3$) type of water, while Quadrant 4, representing BH4, SWu and SWd samples, indicate mixed Calcium Magnesium Chloride (CaMgCl) types of water (Figure 4.4). Similar types of groundwater and surface water around the landfill site were also reported (Makhadi et al., 2020). In terms of cation content, the results showed that the water samples near the Botshabelo dump site were primarily not found in any dominant type (BH3, SWu, and BH4), but an Mg type was also observed (BH2 and SWd). The anion triangle revealed that the BH1, BH3, and BH2 samples had no dominating type, but the other water samples (BH4, SWu, and SWd) were distributed in HCO_3 type of water. All surface water and BH4 were classified as $CaMgHCO_3$, showing that there was little mineralisation regarding main ions, implying that this was fresh water and newly replenished groundwater (Lalumbe and Kanyerere, 2022).

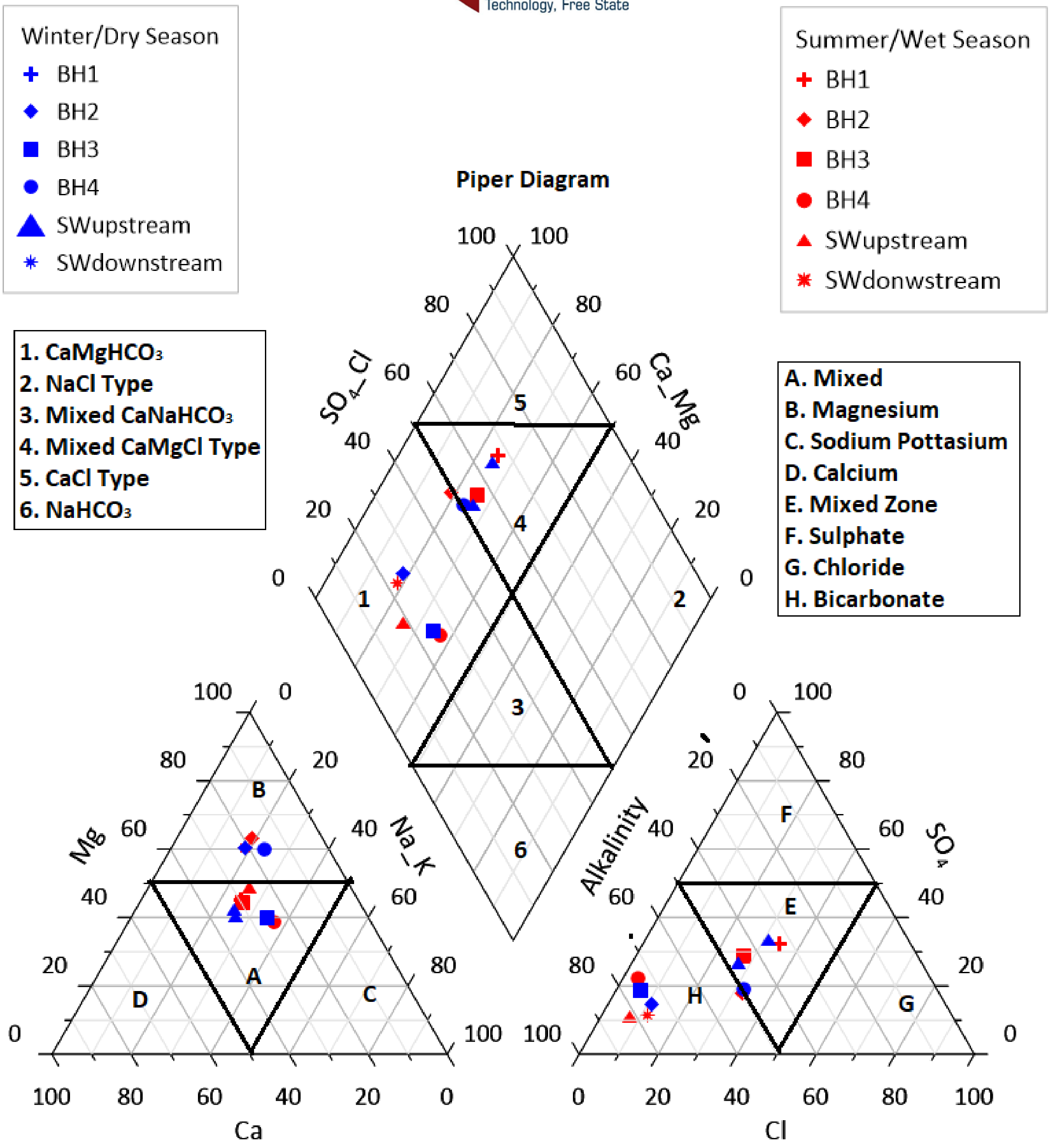


Figure 4.4 ***Piper trilinear diagram for groundwater and surface water in both seasons*** *(Designed by author using Grapher Golden Software. Classification adopted from Jadon et al., 2024)*

The water chemistry data from the Botshabelo non-engineered landfill site revealed that the majority of groundwater and surface water was CaMgCl and $CaMgHCO_3$, Groundwater and surface water chemistry are controlled by separate processes and mechanisms. Climate, water−rock interactions, precipitation, and evaporation all have the potential to significantly affect groundwater and surface water chemistry (Saha et al., 2019). The hydrochemical facies results revealed that the majority of the samples collected near the Botshabelo landfill site fell into no dominant type of water, with some Mg type for cations and bicarbonate for anions, implying the possibility of carbonate and silicate mineral dissolution/weathering and ion exchange processes in groundwater and surface water at various stages (Lalumbe and Kanyerere, 2022). The presence and activity of organic matter in the aquifer causes carbon dioxide to be converted to bicarbonate under oxidative circumstances (Saha et al., 2019). Furthermore, in addition to natural processes, human activities are key external influences that might affect the hydrochemical composition of water (Xiao et al., 2021). Water−rock interaction, evaporation, carbonate weathering, and anthropogenic sources in the research area all have a significant impact on the chemistry of the water bodies.

Because the majority of the groundwater and 50% of the surface water samples had no dominant ions, the Durov diagrams in Figure 4.5 confirmed this phenomenon by showing that all groundwater samples, except BH4, were plotted in the mixing field, indicating quality degradation due to mixing processes. According to several researchers, the SAR accurately predicts the degree to which irrigation water participates in cation-exchange reactions in soil (Makhadi et al., 2020; Ololade et al., 2019; Saha et al., 2019). High levels of SAR indicate a risk of salt replacing absorbed Ca and Mg, resulting in a condition that eventually destroys the soil structure (Saha et al., 2019).

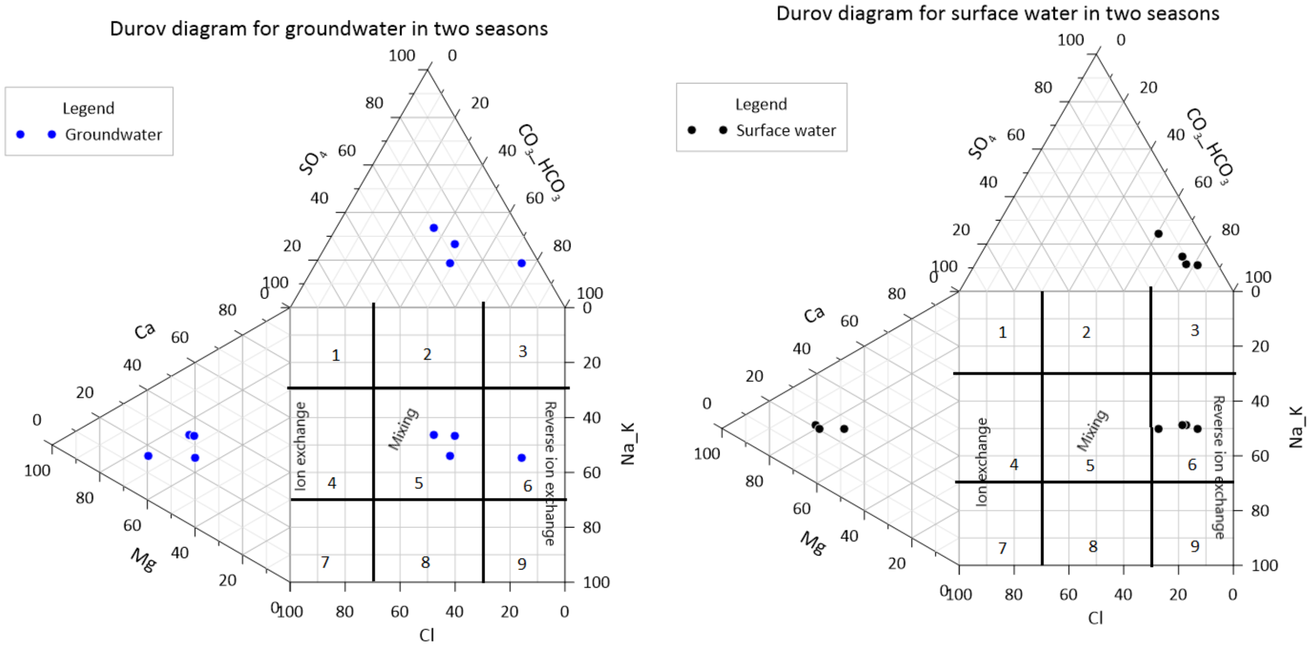


Figure 4.5 ***Durov diagrams for groundwater and surface water for both seasons*** *(Designed by author using Grapher Golden Software)*

4.5 Water quality indices

WQIs are used to determine the overall water quality of sampling places in order to better understand the parameters that affect the water. As previously stated in Chapter 2, water quality indices are used to translate complicated data that is difficult to perceive and comprehend into simple mathematical numerical values (Namugize and Jewitt, 2018). Two WQIs created by Nemerow and Sumitomo (1970), sometimes known as NPI, were employed in this study. Brown et al. (1970) created yet another WQI. These WQIs were combined to determine the water quality at the dump site. Furthermore, employing the two approaches combined improved the reliability of the data and provided additional insight into water quality. Table 4.3 below gives all the classifications of all the water quality indices methods.

Table 4.3 The water quality indices classification for different methods

Index method	Range	Classification	Application	References
NPI	≤1	No pollution		Su et al. (2022)
	1,2	Slight pollution		
	2,3	Light pollution		
	3,5	Moderately polluted		
	>5	Seriously polluted		
WAWQI	0−25	Excellent water	Drinking, irrigation, industrial	Brown et al. (1970)
	25−50	Good water	Drinking, irrigation, industrial	
	50−75	Poor water	Irrigation, industrial	
	75−100	Very poor water	Irrigation	
	>100	Water unsuitable for drinking	Proper treatment before use	
NSFWQI	Below 50	Excellent		Geethamani et al. (2023)
	50−100	Good		
	101−200	Poor		
	201−300	Very poor		
	Above 300	Unsuitable for drinking		

4.5.1 Nemerow's pollution index

The NPI emphasises the most polluting parameter as a primary consideration (Su et al., 2022). It is a single-factor index that determines water quality by computing an overall pollution index, which is a weighted WQI that takes into account elevated values. The NPI approach has the benefit of being rapid and straightforward to utilise, in comparison to other methods. The pollution index values in the table are used to classify the pollution levels of both surface water and groundwater from the landfill site by means of ranges, whereby the NPI values of less than 1 showed that parameters have no influence in polluting the water, while the NPI values greater than 1 denoted the pollution potential of the parameters on water.

Tables 4.4 and 4.5 exhibit the NPI results for groundwater and surface water, displaying the polluting parameters for both seasons. In the tables, the NPI is utilised in reference to the WHO and DWAF requirements for drinking and irrigation compatibility.

Mg, Na, Mn, Mo, and U, as well as EC, polluted groundwater near the dump site in both wet and dry seasons, exceeding WHO and DWAF standards with NPI values of more than one. In the rainy season, the average NPI values for EC, Mg, Mn, and U were 1,76, 1,05, 3,00, and 2,04, respectively. In the dry season. Only EC and Mn had averages of 1,73 and 1,67. In the wet season, the groundwater was both slightly and lightly polluted, whereas in the dry season it was only lightly polluted. The irrigation water quality based on the DWAF regulations revealed computed average NPI values of 12,01 for Mn and 3,89 for Mo and 6,11 for U, which classified the groundwater as seriously polluted for irrigation purposes in the wet season. The dry season also yielded mean NPI values of 1,04, 1,20, and 12,58 for salt, B, and Mn, categorising the water as mild to severely polluted. The boreholes at the dump site showed higher and more serious pollution from trace elements than from main elements. According to Foko et al. (2023), landfill sites are a source of heavy trace metals that enter groundwater via leachate.

Table 4.4 The mean calculated values of the Nemerow Pollution Index for groundwater based on drinking and irrigation quality over wet and dry seasons

Parameter	Wet season			Dry season			Wet season			Dry season		
	WHO			WHO			DWAF			DWAF		
	Min	Max	Mean	Min	Max	Mean	Min	Max	Mean	Min	Max	Mean
pH	0,94	0,94	0,94	0,94	0,94	0,94	0,95	0,95	0,95	0,95	0,95	0,95
EC	0,29	0,64	0,46	0,29	0,62	0,45	1,12	2,40	**1,76**	1,12	2,34	**1,73**
TDS	0,66	0,96	0,81	0,61	0,96	0,78	–	–	–	–	–	–
Ca	0,61	1,21	0,88	0,58	1,30	0,65	–	–	–	–	–	–
Mg	0,85	1,22	**1,05**	1,02	1,17	0,67	–	–	–	–	–	–
Na	0,25	0,52	0,37	0,21	0,57	0,39	0,31	1,51	0,81	0,62	1,30	**1,04**
K	0,00	0,02	0,01	0,01	0,02	0,01	–	–	–	0,00	0,00	0,00
Cl	0,00	0,24	0,12	0,04	0,49	0,31	0,02	1,31	0,56	0,12	1,49	**1,00**
NO_3^-	0,00	0,01	0,05	0,00	0,02	0,01	–	–	–	–	–	–
NO_2^-	0,00	0,01	0,00	0,00	0,00	0,00	–	–	–	–	–	–
SO_4	0,03	0,13	0,07	0,03	0,13	0,07	–	–	–	–	–	–
Al	0,62	0,99	0,81	0,10	0,20	0,12	0,00	0,80	0,35	0,00	0,00	0,00
as	0,02	0,02	0,02	0,00	0,00	0,00	–	–	–	–	–	–
Sb	0,03	0,03	0,03	0,00	0,00	0,00	–	–	–	–	–	–
Ba	0,02	0,07	0,04	0,04	0,11	0,07	–	–	–	–	–	–
B	0,01	0,02	0,01	0,02	0,02	0,02	0,01	0,86	0,30	1,20	1,20	**1,20**
Cd	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00
Co	–	–	–	–	–	–	0,00	0,00	0,00	0,00	0,00	0,00
Cr	0,15	0,40	0,31	0,02	0,04	0,02	0,07	0,20	0,15	0,01	0,02	0,01
Cu	0,00	0,03	0,01	0,00	0,00	0,00	0,01	0,34	0,09	0,05	0,05	0,05

Parameter	Wet season			Dry season			Wet season			Dry season		
	WHO			WHO			DWAF			DWAF		
	Min	Max	Mean	Min	Max	Mean	Min	Max	Mean	Min	Max	Mean
Fe	–	–	–	0,00	0,00	0,00	0,00	0,01	0,01	0,00	0,00	0,00
Mn	0,01	10,70	**3,00**	0,02	6,25	**1,67**	0,05	42,85	**12,01**	0,10	25,00	**12,58**
Ni	0,00	0,00	0,00	0,01	0,03	0,02	0,00	0,00	0,00	0,01	0,01	0,01
Mo	–	–	–	–	–	–	0,64	12,07	**3,89**	0,10	1,20	0,70
Pb	0,00	0,04	0,01	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00
Se	0,00	0,00	0,00	0,00	0,02	0,02	0,01	0,01	0,01	0,00	0,05	0,02
U	0,48	3,80	**2,04**	0,06	0,33	0,21	1,45	11,40	**6,11**	0,20	1,00	0,80
V	–	–	–	–	–	–	0,00	0,73	0,18	0,01	0,10	0,04
Zn	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,01	0,00	0,01	0,01	0,01
F	0,10	0,25	0,19	0,17	0,25	0,18	0,13	0,19	0,14	0,11	0,12	0,11

Table 4.5 The mean calculated values of the Nemerow Pollution Index for surface water based on drinking and irrigation over wet and dry seasons

Parameter	Wet season			Dry season			Wet season			Dry season	
	WHO			WHO			DWAF			DWAF	
	Min	Max	Mean	Min	Max	Mean	Min	Max	Mean	Min	Max
pH	0,94	0,94	0,94	0,94	0,94	0,94	0,95	0,95	0,95	0,95	0,95
EC	0,18	0,22	0,2	0,18	0,22	0,2	0,69	0,82	0,755	0,69	0,84
TDS	0,28	0,33	0,305	0,28	0,33	0,305	–	–	–	–	–
Ca	0,402	0,406	0,404	0,374	0,38	0,377	–	–	–		
Mg	0,414	0,473	0,4435	–	–	–	–	–	–	–	–
Na	0,081	0,097	0,089	0,082	0,131	0,1065	0,232	0,279	0,2555	–	–
K	0,007	0,0135	0,01025	0,017	0,022	0,0195	–	–	–	0,234	0,374

Parameter	Wet season			Dry season			Wet season			Dry season	
	WHO			WHO			DWAF			DWAF	
	Min	Max	Mean	Min	Max	Mean	Min	Max	Mean	Min	Max
Cl	0,022	0,025	0,0235	0,045	0,069	0,057	0,067	0,075	0,071	–	–
NO_3^-	0,003	0,003	0,003	0,003	0,003	0,003	–	–	–	0,134	0,207
NO_2^-	0,002	0,004	0,003	0,003	0,003	0,003	–	–	–	–	–
SO_4^-	0,009	0,0104	0,0097	0,015	0,026	0,0205	–	–	–	–	–
Al	0,49	0,58	0,535	0,2	1,3	0,75	0,009	0,011	0,01	–	–
as	0,022	0,022	0,022	0	0	0	0,002	0,002	0,002	0,004	0,026
Sb	0,032	0,032	0,032	0	0	0	–	–	–	–	–
Ba	0,027	0,041	0,034	0,038	0,108	0,073	–	–	–	–	–
B	0,007	0,008	0,0075	0,017	0,017	0,017	0,034	0,04	0,037	–	–
Cd	0,003	0,003	0,003	0	0	0	0,001	0,001	0,001	0,8	0,8
Co	–	–	–	–	–	–	0	0	0	0	0
Cr	0,012	0,012	0,012	0,04	0,04	0,04	0,006	0,006	0,006	0	0
Cu	0,013	0,063	0,038	0,005	0,005	0,005	0,135	0,635	0,385	0,02	0,02
Fe	–	–	–	–	–	–	0,01	0,02	0,015	0,05	0,05
Mn	0,337	1,587	0,962	0,075	0,125	0,1	1,35	6,35	**3,85**	0,003	0,02
Ni	0,001	0,001	0,001	0,029	0,043	0,036	0,034	0,04	0,037	0,3	0,5
Mo	–	–	–	0	0	0	0,012	0,09	0,051	0,01	0,015
Pb	0,004	0,004	0,004	0	0	0	0	0	0	0,1	0,2
Se	0,004	0,0042	0,0041	0	0,025	0,0125	0,008	0,008	0,008	0	0
U	0	0	0	0	0,1	0,05	0,001	0,001	0,001	0	0,05
V	–	–	–	–	–	–	0,01	0,471	0,2405	0	0,3
Zn	0,001	0,001	0,001	0,003	0,003	0,003	0,004	0,004	0,004	0,01	0,1

Parameter	Wet season			Dry season			Wet season			Dry season	
	WHO			WHO			DWAF			DWAF	
	Min	Max	Mean	Min	Max	Mean	Min	Max	Mean	Min	Max
F	0,14	0,153	0,1465	0,105	0,115	0,11	0,105	0,115	0,11	0,1	0,24

Lukasik and Dabrowska. (2022) used the NPI to assess a waste site in Dab Gornicza in Poland. They investigated seven parameters, some of which were identical to the current study, including Cl, SO_4, B, and Fe, while others were distinct. Their findings showed that the average NPI values for most of the piezometers except ammonia dropped below the NPI threshold of one, indicating a low danger of groundwater pollution by the parameters. They concluded that water chemistry has a direct impact on NPI levels. Similar to the results of the current investigation, NPI values surpassed the limit of 1,0, suggesting pollution potential/risk to groundwater by Ca, Mg, Cl, Mn, Mo, and U. The NPI correlates with the chemistry of the parameters which exceeded both the WHO and DWAF limits for drinking and irrigation, with the exception to boron that showed no pollution based on the water quality results. Similar observations to this study were made by Lalik and Dabrowska (2024), who studied the groundwater status of the landfill in Chorzow, Southern Poland. They analysed for EC, ammonium ions, Cl, nitrate, SO_4, and N. They found that the major ions posed significant risk to groundwater pollution, with excess NPI values exceeding 40, which was related to their elevated chemical concentrations and showing impact by the landfill site. Therefore, the groundwater for the present study will pose risk for both drinking and irrigation purposes upon usage due to its deteriorated quality by the landfill site. The surface water revealed no pollution on the majority of the parameters except Mn, which indicated moderate pollution for irrigation quality in the rainy season with a mean NPI value of 3,89, whereas no pollution was observed during the dry season for both drinking and irrigation purposes. This contradicts the study's surface water quality results, which showed that Mn did not pollute surface water. Furthermore, Alam et al. (2021) conducted a research on the surface water quality at a landfill in Sylhet, Bangladesh, where they discovered different results than those of this study. Their findings indicated deteriorating surface water quality due to the landfill site that was backed up by results of the NPI. Amano et al. (2020) supported the findings of Alam et al. (2021) on the relationship between water quality and the WQI. Despite this inconsistency, the vast majority of this study's surface WQI measurements aligned and substantially accorded with the findings of the aforementioned research.

4.5.2 Weighted arithmetic water quality index

The WAWQI provides an overall or combined quality of water (Chidiac et al., 2023), it was chosen as the best index to utilise over other methods. Table 4.6 shows the overall average WQI for both drinking water and irrigation.

Table 4.6 The mean calculated values for Weighted Arithmetic Water Quality Index for groundwater and surface water over wet and dry seasons for drinking and irrigation

Wet season	**WHO**			**DWAF**		
Sampling point	**Min**	**Max**	**Mean**	**Min**	**Max**	**Mean**
Groundwater	6,14	43,23	20,43	63,53	922,58	342,64
Surface water	2,53	4,94	3,74	35,29	68,85	52,07
Dry season	**WHO**			**DWAF**		
Sampling point	**Min**	**Max**	**Mean**	**Min**	**Max**	**Mean**
Groundwater	0,96	15,06	5,03	9,63	308,90	162,95
Surface water	0,80	3,21	2,00	7,52	11,67	9,59

The WAWQI readings for groundwater in the wet season averaged 20,43 and 342,64 for drinking and irrigation water quality, respectively. Brown et al. (1970) classified groundwater as excellent for drinking but very poor for irrigation during the wet season. In terms of the dry season, the average values for drinking and irrigation were 5,03 and 162,95, respectively, indicating that it was both good and very poor, comparable to the rainy season. Oni and Fasakin (2016) examined the potability of both groundwater and surface water in the area of a waste site using the same WAWQI. They used 10 parameters of which the majority were similar to this study that incuded pH, Fe, Cl, Pb, Zn, Cu, nitrates, nitrites and total coliforms. Their computed average WQI for groundwater samples was 337, with all groundwater samples designated as unsuitable for drinking due to uncontrolled dumping of both hazardous and non-hazardous waste, as well as surface runoff of leachate. In addition, the shallow water level of 5 m and the lateritic soil allowed for easier contaminant percolation. The current study, however, had water levels that were also shallow at less than 10 m, but the soil cover was clay, the type of garbage dropped is general, non-hazardous, and most of the waste is recycled, even though the landfill itself is not a proper sanitary landfill.

As a result, it presented much lower average WQI values for drinking, while for irrigational use were very high in groundwater. Oni and Fasakin (2016) also used the WAWQI near a landfill site in Nigeria, where they discovered low drinking groundwater quality as a result of lead-containing leachate that entered from the landfill surface and caused pollution. Chilukuri et al. (2019) observed similar findings from a waste site in India. Despite the contamination found in the water quality results, the WAWQI method classed groundwater at the Botshabelo dump site as excellent for drinking, which contradicts the NPI results, which revealed substantial pollution by some metrics. These differences can be attributable to the lack of accuracy of the WQI models.

In the wet season, the average surface water WAWQI values were 3,74 and 52,07 for drinking and irrigation, respectively, indicating that the water was excellent for drinking but inadequate for irrigation. The dry season saw good surface water quality for both drinking and irrigation, with mean values of 2,00 and 9,59, respectively. These findings were consistent with the surface water quality results, which showed minimal to no contaminating effect of the landfill on surface water quality. Oni and Fasakin (2016) reported poor quality surface water at a landfill due to leachate runoff during rainstorm activity. Menberu et al. (2021) obtained similar results in Hawassa Lake in Ethiopia.

4.5.3 National Sanitation Foundation water quality index

Table 4.7 shows the findings of the WQI of the NSFWQI for both the wet and dry seasons. As stated in chapter 2, this method was frequently utilised to calculate the WQI for various water applications or consumption. Groundwater had mean values of 241, 67 and 255,85 for both the wet and dry seasons, and it was classified as very poor for drinking quality (Table 4.3). The results contradicted those reported by Onukwugha et al. (2020) in a landfill site in Nigeria, which revealed the overall value of 65,76 that was defined as a medium grade of water. In another study, Vashisht et al. (2017) calculated the WQI using the NSFWQI and obtained a value of 52,69, indicating that the water was in good condition. The NSFWQI classification for the estimated values in this study contradicted nearly all of the water quality outcomes, which revealed no pollution of groundwater as a result of the landfill. For example, the study by Vashisht et al. (2017) indicated excess concentrations of parameters such as alkalinity, TDS, total suspended solids, Cl, hardness, and SO_4 that were higher than the limits; however, the water quality according to the NSFWQI categorised the water as good quality. However, the agreement occurs with groundwater pollution, as evidenced by high mean concentrations of Mg, Mn, and U. This pollution could be the cause of the extremely poor drinking water quality in this study. Surface water results calculated using the NSFWQI yielded values of 101,70 and 100,15 for both seasons. The surface water was deemed poor in quality during the wet season, but good in the dry season.

Table 4.7 National Sanitation Foundation Water Quality Index groundwater and surface water over wet and dry seasons

Wet season			
Sampling point	**Min**	**Max**	**Mean**
Groundwater	194,10	319,20	241,67
Surface water	97,50	105,90	101,70
Dry season			
Sampling point	**Min**	**Max**	**Mean**
Groundwater	186,70	334,50	255,85
Surface water	98,80	101,50	100,15

Akharame and Obianke (2024) used the NSFWQI to assess the quality of river water in Benin City, Nigeria. They recorded the WQI range of 54,04 to 61,95 in different seasons both upstream and downstream, which defined as intermediate water quality that required extra treatment before use or consumption. Total coliform, turbidity, phosphate, and pH were polluting parameters caused by anthropogenic activities such as agricultural operations, industrial facilities, and religious activities near rivers. Noori et al. (2018) used non-traditional NSFWQI metrics to determine the river's medium and bad water quality status in both wet and dry seasons. Similar to the water quality results of this study, the good water quality of the surface water corresponded to zero contamination impact of the landfill site on its quality, even though it had a poor quality for the wet season, which could be attributed to runoff induced by high rainfall activity that caused erosion and deposition of soil and other landfill site substances which did not occur during the dry season.

The use of several water quality indices allowed for a clearer and more accurate assessment of the quality of both groundwater and surface water. To emphasise, all the approaches yielded similar classifications for surface water quality: no pollution, outstanding, and good quality for drinking and irrigation, with slight variances. The NPI and WAWQI both concurred that groundwater for irrigation was seriously polluted and of poor quality in both seasons, except the NSFWQI. In terms of drinking quality, both the NSFWQI and the NPI concurred but disagreed with the WAWQI. This will aid in decision-making on water use.

4.6 Groundwater vulnerability

In the realisation of the importance of the protection of groundwater resources from becoming polluted, decision-makers, developers and scientists alike have developed the methods that can predict and discern which areas will more likely be prone to contamination than others as result of human activities. The end goal was that once the areas have been delineated, those that are prone to contamination would be exposed to certain restrictions for usage or even earmarked for in-depth attention. This then led to the inception of groundwater vulnerability or susceptibility which many authors in Chapter 2 have defined. Some authors characterised it in terms of specific pollutant features, while others saw it as a product of human-caused activities on the land surface. Others have interpreted it as a fundamental characteristic of the subsurface media. For the current investigation, the GOD and RTt vulnerability methodologies were employed to determine groundwater vulnerability at the Botshabelo landfill site.

4.6.1 Groundwater vulnerability assessment using the GOD model

The GOD model is an overlay and index approach method that takes into account the occurrence of groundwater (G), overall lithology (O), and groundwater or aquifer depth (D) (Nnandozie et al., 2019). The GOD index was determined by multiplying the ratings assigned to each of the three factors. Table 4.8 presents the GOD index results for the research area. Table 4.9 lists the GOD classes and their respective index values. The GOD index for the present investigation was evaluated at 0,08, as shown in Table 4.8.

- **Groundwater occurrence**

The aquifer type in the Free State and Botshabelo areas was determined using the maps of the DWS (2012a) and Conrad et al. (2019) classification as fractured and intergranular. However, a prior study on the geohydrological assessment of the same location by Van Niekerk (1996) discovered that while drilling, groundwater was located in the restricted sandstone layers that occurred between the impermeable mudstone layers. The groundwater at the present study location is now of no supply for consumption of any kind because they are monitoring boreholes; however, due to the country's depreciating water supply, groundwater remains the future alternative water supply source.

- **Overlying strata**

The study area is overlaid by strata of Supergroup geology in the Beaufort successions that mainly comprises of mudstone, shale and siltstone. The previous study by Van Niekerk (1996) also reported similar geological formations of mudstone that occurs overlying the sandstone strata. Moreover, the study area is characteristic of the clay cover that occurs on the surface, thus, acting as a solid barrier against any emerging and infiltrating contaminants triggered by the landfill site. Furthermore, Van Niekerk (1996) also discovered low recovery rates during permeability or pump tests that were conducted in the landfill site, which signals relatively low permeability.

Table 4.3 The calculated GOD index for groundwater using parameters and ratings

Factor	Range	Rating	Weight	GOD index
Groundwater occurrence/aquifer type	Confined	0,2	–	0,2
Overlying lithology	Mudstone/shale	0,5	–	0,5
Depth to groundwater (m)	5−10	0,8	–	0,8
$\boldsymbol{GOD = G \times O \times D}$ **= 0,08**				

Table 4.4 GOD vulnerability classification

Vulnerability class	Vulnerability index
Very low vulnerability	0–0,1
Low vulnerability	0,1–0,3
Moderate vulnerability	0,3–0,5
High vulnerability	0,5–0,7
Extreme vulnerability	0,7–1,0

Source: Deubalbe et al. (2021)

- **Depth to water table**

The water level in the research area ranged from 3 m to 10 m for all boreholes. The water level was monitored with a Solinst water level meter. The prior investigation discovered the boreholes to be ±30 m deep, indicating that they were previously drilled. The deeper the water level, the smaller the risk for groundwater pollution; conversely, shallow groundwater levels increase the possibility of contamination. This is because pollutants take longer to reach groundwater at deeper water levels (Linggasari et al., 2020).

- **GOD vulnerability map**

The GOD model's parameters were derived and mapped using ArcGIS 10,5 software. It was interpolated using the Krigging interpolation in the geostatistical analyst tool, which generated the raster map and interpolated the data points displayed in Figure 4.6. The GOD map was created by plotting the computed GOD values for roughly 20 borehole sites. The Botshabelo landfill site contained only four boreholes, both inside and beyond its proximity, but the regional borehole data collected from the DWS portal covered a larger area for vulnerability. Table 4.9 displays the GOD index classifications: extremely low vulnerability (0−0,1), low vulnerability (0,1−0,3), moderate vulnerability (0,3−0,5), high vulnerability (0,5−0,7), and extreme vulnerability (0,7−1,0). The groundwater in the current investigation was rated as very low vulnerability at 0,08, as shown in Table 4.8, The very low vulnerability encompasses 100% of the research area, as the groundwater samples have the same vulnerability indices. The aquifer confinement to impermeable musdstone layers coupled with the clay cover on the land surface was the most contributing factor to the low vulnerability despite its shallow water levels. This contradicts the regional vulnerability map plotted by the DWS. DWS (2012b) compiled a map for the overall regional groundwater vulnerablility of South Africa, wherein, for the present study involved, showed moderate vulnerability in the generated map. Moreover, the studies by Huisamen and Burger (2015) on a landfill site in KwaZulu−Natal, and Belle and Saungweme (2020) in the Northern Cape municipal area agreed with the DWS results. The GOD vulnerability distribution of the Botshabelo landfill shows very low vulnerability for all the groundwater monitoring boreholes. Despite the shallow levels of water in the research area, the aquifer's confinement could be the likely source of the very low vulnerability.

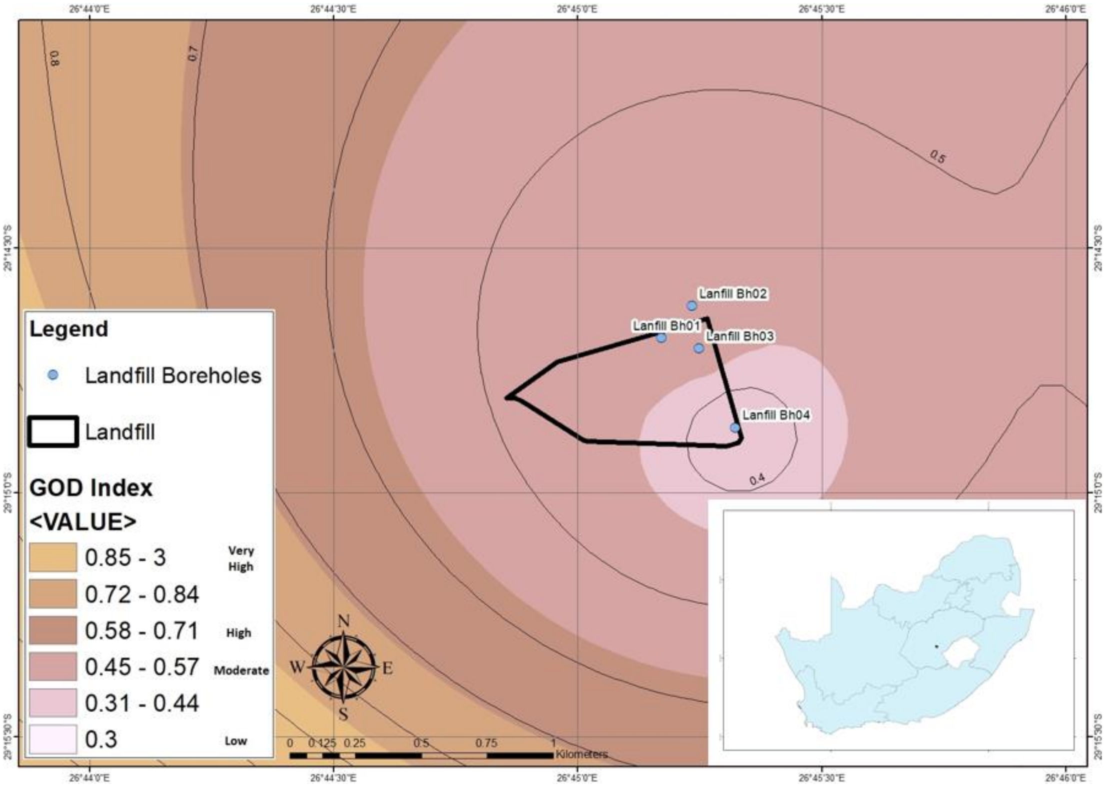


Figure 4.2 GOD index map for groundwater *(Mapped using ArcGIS 10 version)*

4.6.2 Groundwater assessment using the rainfall travel time method

The RTt method is a subjective method that takes into consideration the recharge element which is the rainfall that passes through a specified soil thickness in a vertical movement during the downward infiltration that ultimately recharges the groundwater. It is a source−pathway−target method that was developed by Oke (2016), which is computed by multiplying the three parameters of the RTt as shown in Chapter 2. Similar to the GOD model, it uses the weighting approach for its various parameters.

- **Rainfall factor**

The R-factor is based on the amount of rainfall in a specific area. According to Oke (2016), the R-factor posits that the more the rainfall in a given area, the higher the rate of infiltration, and thus the greater the aquifer recharge. However, this does not account for the soil medium or slope qualities of the terrain through which percolating rainfall occurs. The research region is located in an area characterised by summer rainfall and defined as a sub-humid, warm zone with water scarcity (Mangaung Metropolitan Municipality, 2020). Furthermore, Botshabelo receives an annual rainfall of around 323 mm in summer time and 85 mm in winter time. Furthermore, Botshabelo has an annual rainfall of approximately 323 mm in summer and 85 mm in winter and an annual rainfall of 533 mm per year, in comparison to the country's annual precipitation of 434 mm per year. In contrast, the Skhemelele landfill site had an annual rainfall of 1 025 mm per year as it is located in the KwaZulu−Natal area with torrential rains and thunderstorms. Also, the study conducted by Oke (2016) reported high precipitation rates of between 1 200 mm and 1 800 mm per year, which led to the higher potential of infiltration rates than the present study, which may suggest low rates of infiltration. High surface flows due to rainfall often result in the formation of drainage pathways that have a direct connection to the aquifer thus increasing its rate of recharge. The weight of the annual rainfall is given in Table 4.10.

Table 4.5 The calculated RTt index for groundwater using weights and parameters

Parameter	Range	Weight
Rainfall	800−400 mm	2
Overall depth and Slope (m)	Depth (5−10 m) Slope (5−25 m)	2,0 and 1,2
Porosity (Soil type and Rock type)	Clay, shale and igneous rock	5 and 1
Hydraulic conductivity	Massive clay Dense rock, shale, Ign. and meta rocks Clay	5
$RTt = R \times (10) + (D \times S \div (K_{sat} / \Theta)) \times (10) = 52$ **Average RTt Index is equal to 52**		

- **Travel time factor**

Oke (2016) defined travel time as the rate at which a percolating fluid flows through the strata (unsaturated zone) until it reaches the groundwater table. In addition, the transit time represents the channel along which the infiltrating liquid travels, which is mostly determined by the lithological units and soil characteristics. The geology of the area is composed of the mudstone and clay units which both have high permeability and low porosity to prevent the absolute contaminant infiltration into groundwater, and result in low vulnerability. Similar to the GOD index, the water level depth in the RTt is considered as it signifies the thickness of the lithology. The study area was composed of water levels between 4 m and 10 m, which represented shallow water levels as measured in the field. The overall depth to water weighed 2,0, as shown in Table 4.11. The slope data was obtained from Google Earth and calculated to produce a slope range of 5−25 m with a weight of 1,2. The slope or topography effects surface runoff, which directly affects the infiltration rate (Bera et al., 2021). The slope indicates where contaminants are likely to settle, percolate, and pollute the underground water. Hence, the greater the slope, the lower the rate of infiltration there is, thus, the lower the contamination ensues. The slope of the Botshabelo landfill site was not flat as it is located next to the Pirana Mountain, and therefore, it was more steeper towards the mountaneous side. Bera et al. (2021) reported variable slope values in the basin where they asserted high conamination risk on the side of the basin that had a low slope as compared to a much steeper slope that had low vulnerability to pollution due to high runoff. Hydraulic conductivity is one of the determinants of contamination in groundwater. The elevated conductivity concentrations in water symbolises contamination, and as a result, an aquifer that has a high flow conductivity leads to a greater risk of groundwater contamination (Assefa and Dinka, 2023). The rock conductivity for the RTt method was chosen where various rock and soil types as well as their textural properties were assigned their hydraulic conductivity values. From Table 4.11, the soil and rock flow conductivities were very low due to low permeability of both massive clay exposure of groundwater to contamination from emerging pollutants. Therefore, a weighting of 5 was given for hydraulic conductivity.

Table 4.6 RTt index classification

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

Source: Oke et al. (2016)

- **Rainfall travel time vulnerability map**

The RTt model was also plotted with ArcGIS 10,5 software’s Inverse Distance Weighted (IDW) interpolation in the Geostatistical Analyst tool (Figure 4.7). The RTt model is likewise a weighted method that employs various parameters that are calculated using the formula to obtain the overall index, as described in Chapter 3. Table 4.11 displays the vulnerability classification of the RTt model, which varies from very low (12−29) to low (29−47), moderate (47−65), high (65−83), and very high (83−100). The Botshabelo landfill vulnerability class had an overall moderate vulnerability of 53 for groundwater, covering 100% of the research region, identical to the GOD model as shown in Figure 4.7. Low rainfall in both the research region and the province, together with low porosity, moderate slope, and hydraulic conductivity, were the most critical elements that contributed to its moderate sensitivity value. This meant that the quantity of the contaminants was relatively low and could not even reach the groundwater table, which, in turn, led to lower chances of groundwater pollution. High or very high vulnerability was not recorded on any side of the landfill even though the water levels for the boreholes were at very shallow levels.

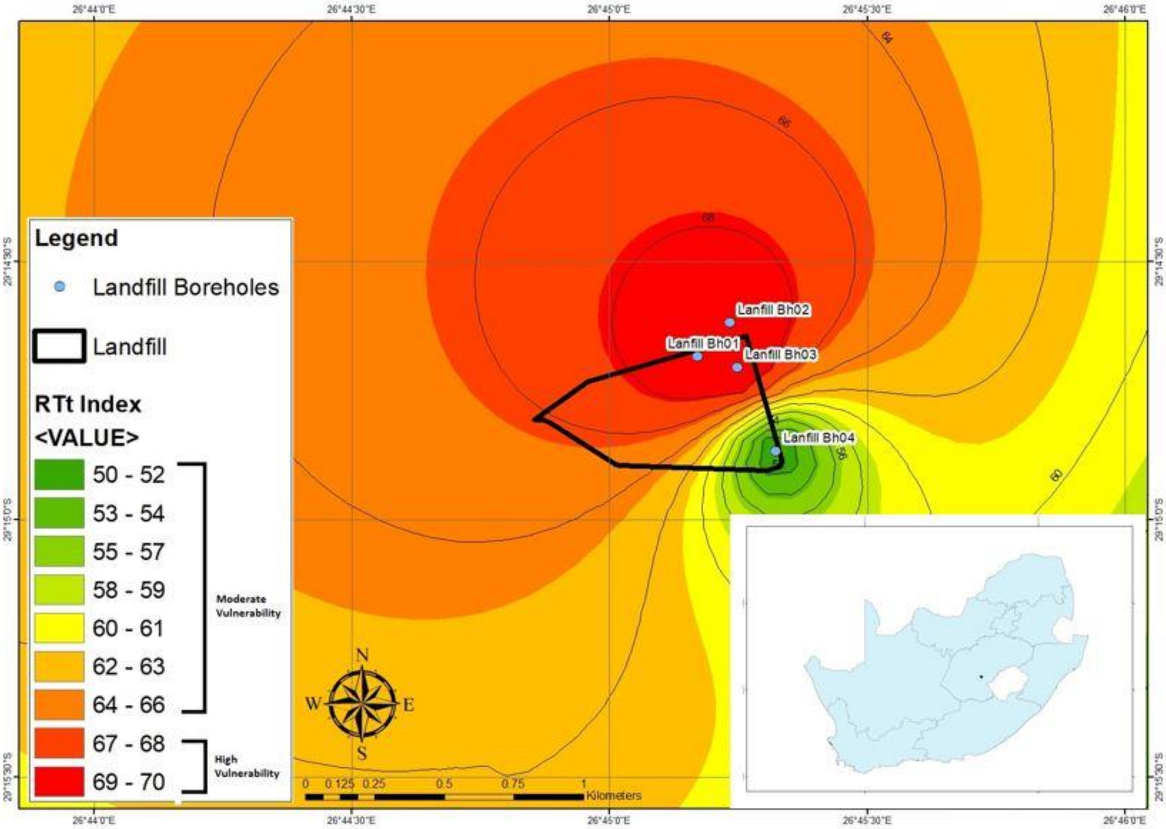


Figure 4.3 ***Rainfall travel time map for groundwater*** *(Mapped using ArcGIS 10 version)*

4.7 Groundwater validation of the GOD and RTt methods

The vulnerability validation method is a way in which the calculated vulnerability indices are compared with the groundwater parameter results for a specific area. This helps the decision-makers and authorities to make sound decisions when selecting sites for a development of land-use activities which might pose more risk to groundwater in an area, and to also delineate if the groundwater is currently at risk or not. Several researchers have conducted a validation process in their groundwater vulnerability studies using various models (Ghazavi and Ebrahimi, 2015; Jhariya, 2019; Oke, 2016). By comparing the vulnerability indices and the water quality parameters it is simpler to discern highly vulnerable areas by observing the concentration of the parameter associated with the degree of vulnerability where high vulnerability can occur due to a high concentration of the immediate contaminating parameter and vice versa. Moreover, groundwater validation helps to discern the most polluting parameter in a case where various parameters are used for validation. For many studies, NO_3^- has often been used as a validation parameter due to its low presence in groundwater whereby any elevation in its concentration can be acutely associated with anthropogenic activities. The present study used several parameters such as EC, NO_3^-, and trace elements due to their abundace in landfill sites in comparison with both GOD and RTt computed vulnerability indices on the correlation plot shown in Figure 4.8. They are plotted on the logarithmic graph using Microsoft Excel on the same graph.

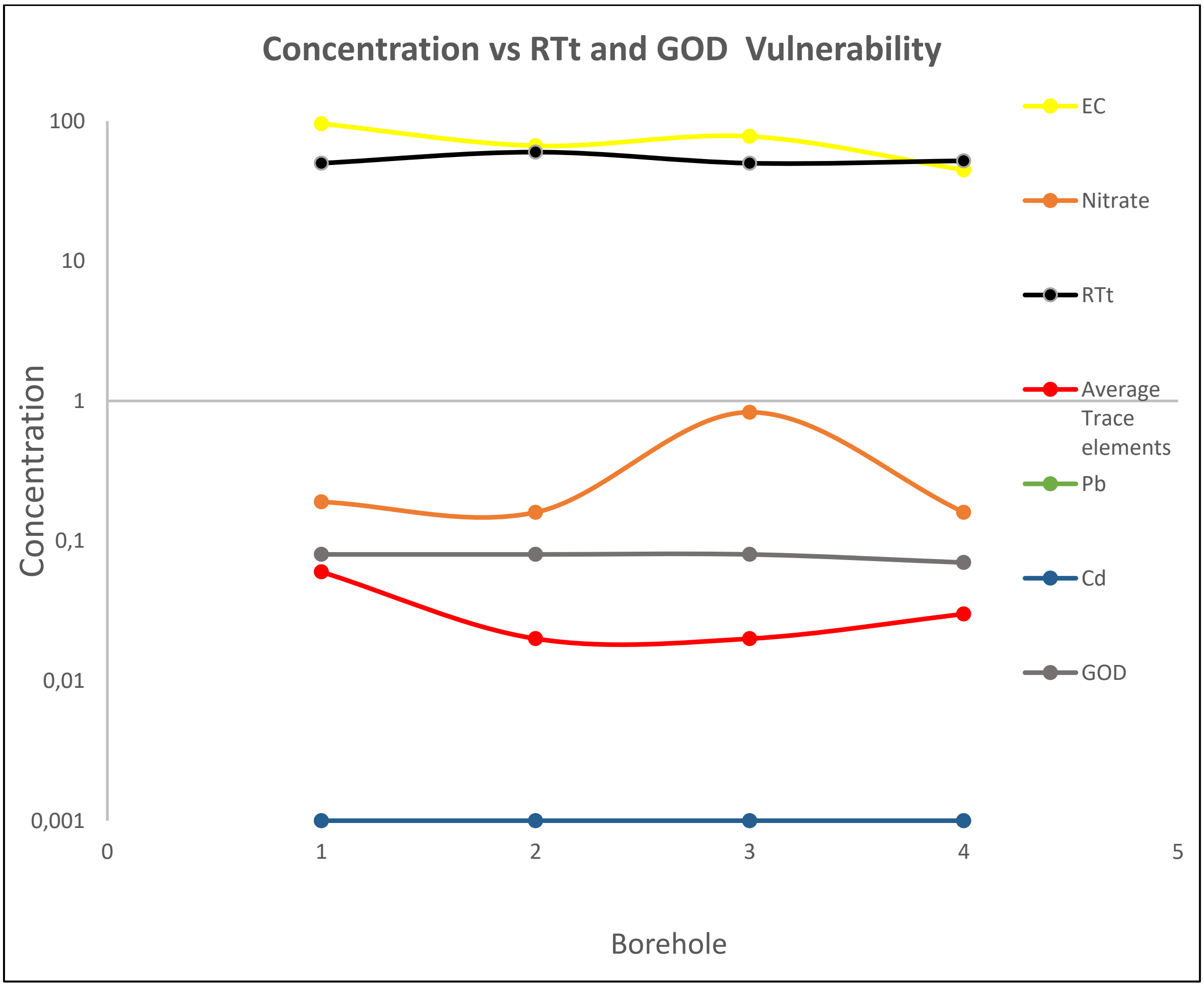
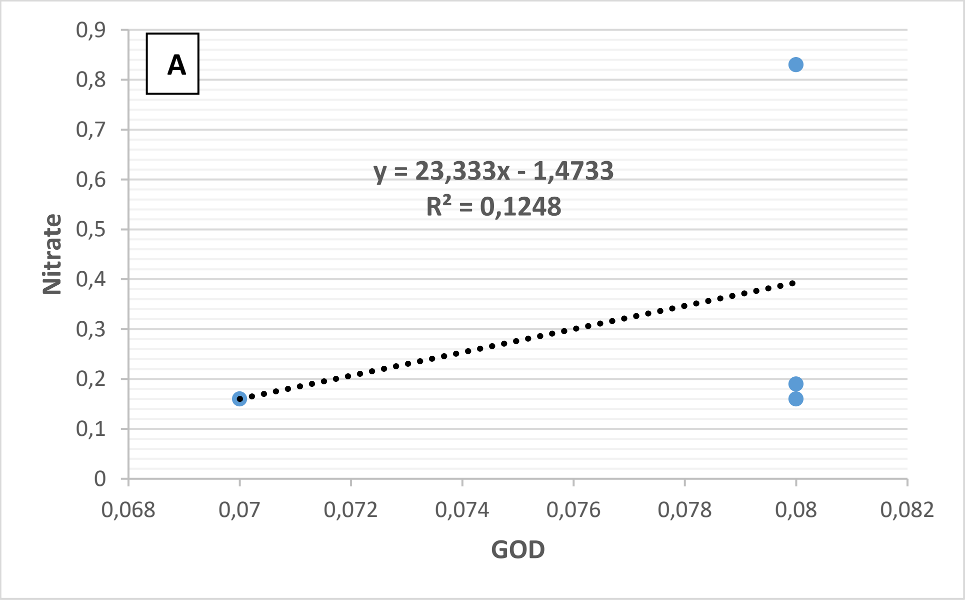


Figure 4.8 Correlation plots for selected parameters versus vulnerability methods

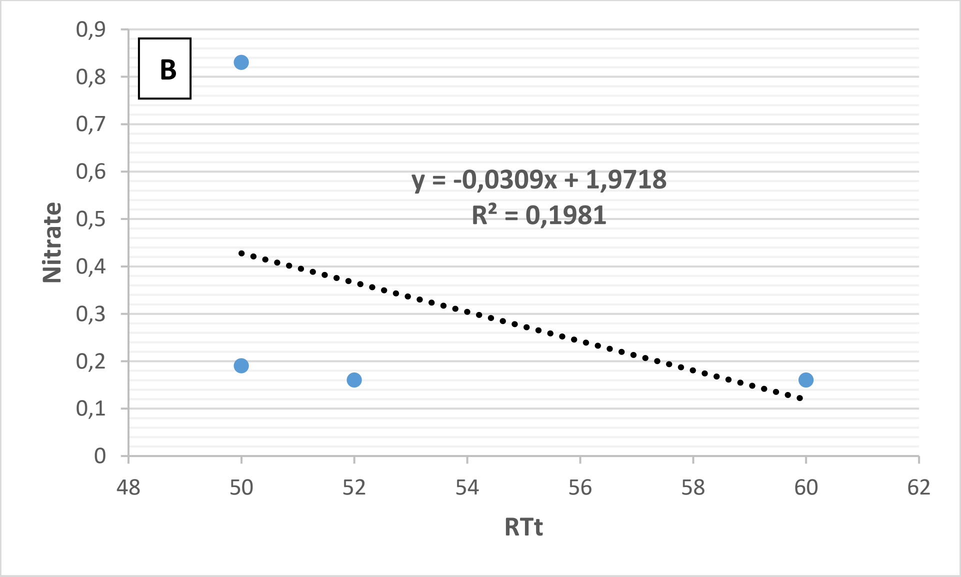
4.7.1 Validation with nitrate

The NO_3^- validation method is the most widely preferred for validation by numerous researchers. This is due to the fact that man-made activities are associated with the contamination of NO_3^- in groundwater resources. Agricultural activities linked with fertilisers can release NO_3^- into groundwater resources due to leaching by rainfall as it recharges from the surface (Jhariya, 2019). NO_3^- is more stable in the oxidation state than in reducing conditions. As stated above, it has relatively low concentrations in groundwater, and as a result, any high concentrations may be attributed to anthropogenic activities, including waste disposal sites. The research area is located in an agricultural zone, and waste disposal from domestic wastes are potential sources of NO_3^- input in the landfill site. In Figure 4.8, the correlation plot of the concentrations of parameters are plotted against the intrinsic vulnerabilities of GOD and RTt for the four boreholes, respectively. The GOD index shows relatively constant index values in all the boreholes in the study area that fell in the low vulnerability class, while the NO_3^- concentrations plotted close to the GOD index as they were found at low concentrations that posed no contamination to groundwater. The NO_3^- plot showed variable concentrations where one borehole had a higher concentration than the rest. This variability had no effect on the vulnerability of the boreholes as it remained equal. The linear regression plot in Figure 4.9a depicted a positive correlation for GOD and nitrate. Low concentrations of NO_3^- for all the boreholes were below the standards of WHO, SANS, and DWAF for drinking and irrigation. The measured mean values were only 0,33 mg/ℓ for both the wet and dry seasons. According to Ghazavi and Ebrahimi (2015), values of more than 10 mg/ℓ indicate anthropogenic pollution. As a result, the NO_3^- in the landfill had minimal concentrations, which corresponded to extremely low vulnerability indices. This is due to the fact that the study region is located in the low vulnerability zone, which is defined by a limited aquifer system that is entirely impermeable to developing contaminants as previously indicated. As a result, the GOD model provided a useful understanding of the relationship between vulnerability classes and borehole NO_3^- concentration.

The RTt model is underpinned on the assumption that areas with low vulnerabilities are often typical of low recharge and high residence time (Oke, 2016). From the correlation plot, the RTt vulnerability increases with decreasing NO_3^- concentrations and vice versa. This shows an inverse relationship (Figure 4.8). This is supported by the linear regression plot in Figure 4.9b. The lower vulnerability in boreholes corresponded with higher NO_3^- concentrations, and this may be due to longer time residence in the vadose zone, which means that NO_3^- is trapped within the rock layers and poses low vulnerability to groundwater due to the landfill leachate. Despite fluctuating NO_3^- amounts in the boreholes, the landfill site is classed as moderately vulnerable, which can be attributed to the time it takes for contaminants to reach groundwater. As previously indicated, the research region is made up of shallow water levels at depths less than 10 m, implying that the bed thickness is substantially lower and contaminants can travel a shorter distance to reach groundwater.



A. Positive regression



B. Negative regression

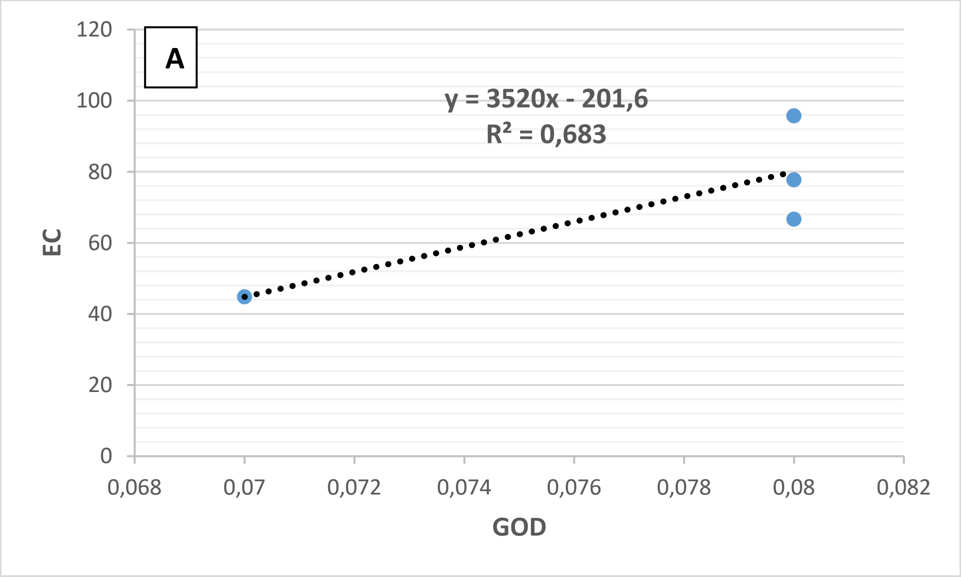
Figure 4.9 Linear regression plots for nitrate validation

4.7.2 Validation with electrical conductivity

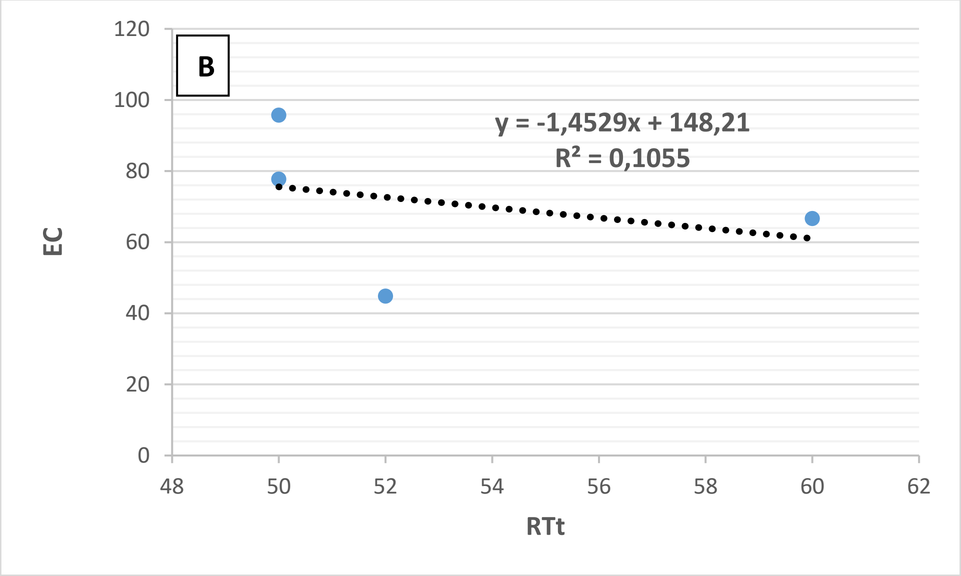
EC is one of the most important groundwater quality parameters that can immediately identify any type of contamination by a percolating fluid medium that reaches the water table. It is determined by the amount and volume of inorganic ionic components and salts that are present in a fluid (Rashid et al., 2022). The landfill has a direct impact on the amounts of inorganic ions and salts that leach into the precipitation that comes into contact with the waste material in the landfill site. Classic general, non-hazardous landfill sites are often obligated to receive organic wastes that predominantly contain ions and salts from household, municipal, construction, demolition wastes that do not carry numerous toxic substances that damage the groundwater quality. However, the likelihood of contaminated fluid in reaching groundwater sources is dependent on the geological, hydrogeological, and the groundwater evolution processes (Przydatek and Kanownik, 2021). The EC in the plot shows slightly variable concentrations that show little change in all four boreholes with the GOD index having constant concentrations. Both plots show rather linear alignment which shows a positive correlation that can be supported by the fact that very low vulnerability is due to low conductivity concentrations in groundwater in the landfill site (Figure 4.8). The physicochemical parameter data revealed that all boreholes had electrical EC concentrations that were below the WHO and SANS permitted levels, but exceeded the DWAF limits throughout both wet and dry seasons. The mean concentrations for both seasons were 95,7 milliSiemens per metre (mS/m), 66,3 mS/m, 77,7 mS/m, and 44,8 mS/m for BH1, BH2, BH3, and BH4, which were all less than the 150 mS/m recommended by the WHO and SANS but greater than the 40 mS/m recommended by DWAF.

The close correlation between the GOD index and EC is also supported by the confined aquifer system that is composed of the impermeable mudstone layers as well as the overlying clay cover that acts a protective cap against downward infiltration of contaminants that pose a pollution risk to groundwater. This increases the residence time of the contaminant in the strata and reduces vulnerability despite the shallow water levels, although Musgrove et al. (2023) reported longer residence times that were associated with deep aquifer levels.

Comparing with the RTt vulnerability index graph, similar obvervations as the NO_3^- can be seen regarding its correlation behaviour. From the correlation and linear regression plots, EC shows a negative correlation with the RTt index, indicating that increasing vulnerability causes a decrease in EC and vice versa (Figures 4.10b). This is also shown by the nature of the pattern of the two plots. Similar to NO_3^- concentrations, elevated EC concentrations are typical of longer residence times in the vadoze zones where infiltrating liquid medium is trapped in the clay lithological units in the landfill site (Van Niekerk, 1996). The vadoze zone in the landfill consists of low porosity of mudstone and clay units which prevent percolation. Furthermore, the hydraulic conductivity of the material is determined by its capacity to let fluid flow through its pores and cracks. The flow conductivity of a given area is determined by the types of rocks present (Saravanan et al., 2019). Therefore, permeable rocks contain high hydraulic condictivity compared to impermeable rocks; hence, an aquifer that contains high flow conductivity tends to pose a high vulnerability to groundwater because it exposes it to incoming contaminants (Assefa and Dinka, 2023). For the present study, the local rocks contain low flow conductivity which thwart the permeation of contaminants and result in low vulnerability. Moderate vulnerability may be associated with shorter residence times in the vadose zone due to shorter flow distances as a result of shallow water levels. If groundwater levels are near the surface it has more risk of contamination due to its proximity to the contaminant source.



A. Positive linear regression



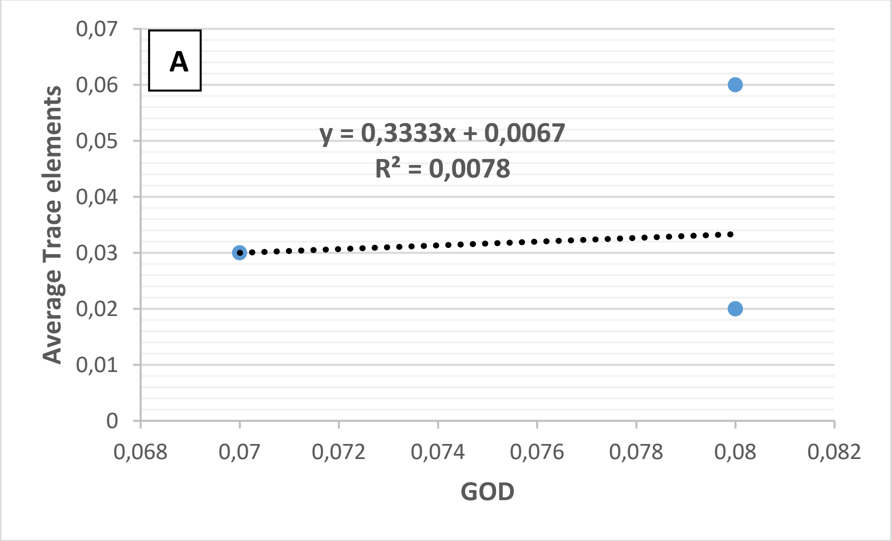
B. Negative linear regression

Figure 4.10 Linear regression plots for electrical conductivity validation

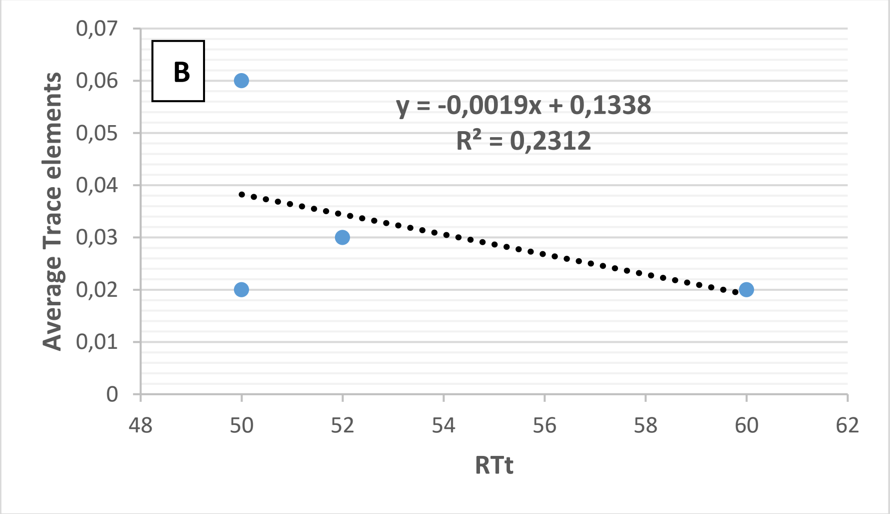
4.7.3 Validation with trace elements

Validating vulnerability with trace elements is very important based on the fact that trace elements tend to seriously pollute and deteriorate the water quality in an extreme manner such that even in trace quantities the water becomes unsuitable for use. Although numerous studies employed NO_3^- to evaluate groundwater susceptibility, very few studies, such as that by Khafouri et al. (2023), used trace elements to validate the intrinsic vulnerability of groundwater to pollution. Validation using trace elements was crucial for the present study due to the fact that waste disposal sites are among the direct sources of numerous trace elements into the environment as many wastes contain these elements in abundance. The correlation plot in Figure 4.8 shows the average 18-trace element plot for all the trace elements during the entire two seasons plus the individual elements plotted against vulnerabilities. Since GOD remained the same over two seasons, it can be noted that the average trace element plot showed slight variability across all the boreholes. However, there is a modest positive association between average trace elements and the GOD index, such that concentrations expressed a slight shift between boreholes. This means that very low vulnerability corresponds with low average trace element values in all boreholes, and that there was little risk posed by trace elements on groundwater (Figure 4.11a). The linear regression also corroborates with correlation plots. From the water quality results reported previously, the majority of the trace elements showed concentrations that were below the drinking and irrigation standards, and also posed no risk to pollution. Deleterious trace elements like As Cd and Pb were all beneath detection limits, and according to Brindha et al. (2020), these elements are a major public concern due to their high persistence in the environment, toxicity, and bio-accumulative properties. The plots of Cd and Pb perfectly correspond with the GOD index to signify very low vulnerability and non-detection in the landfill site. Both of these elements have the capacity to contaminate groundwater even in trace elements due to their high toxicity. Also, the confinement of the aquifer and overlying lithology of the GOD method may have been the probable causes of the very low vulnerability despite the shallow water level depths of the boreholes. Cd and Pb are the most common trace elements found in landfills because to their widespread use in household goods such as batteries, paints, pipes, and photoelectric products that end up in landfills (Nagarajan et al. 2012). Table 4.12 below shows the correlation between water quality parameters and vulnerability methods.

The correlation plots of the RTt index versus average trace elements, Pb, and Cd, also showed an overall positive correlation although there was a slight negative correlation for BH2 over the course of two seasons, while Pb and Cd showed constant concentrations in all boreholes. The moderate vulnerability corresponded with low average concentrations of trace elements, Pb as well as Cd in the dump site, except for BH2, which displays an increasing vulnerability with decreasing trace element concentrations. The linear regression plot, however, showed a negative correlation.T he rainfall factor of the RTt index is the main driver of the contamination degree of groundwater. Rainfall is the migration factor of trace elements from the waste into groundwater through leachate because they do not dissolve or decompose. Soil sorption ability, water movement in soil, rate of reaction of trace elements with solid phase as well as their initial concentration also influence their migration (Nagarajan et al., 2012). As previously stated, clays have high sorption capacity due to their low permeability and porosity, thus, they tend to be attached to trace elements to lenghten their residence time and lower groundwater vulnerability (Oke, 2016). The overall mild or moderate RTt vulnerability agrees with the low amounts of trace elements in groundwater and suggests that no contamination effect was caused by the landfill.



A. Positive linear regression



B. Negative linear regression

Figure 4.11 Linear regression plots for trace element validation

Table 4.7 Mean physicochemical parameters for groundwater and their resultant GOD and RTt indices

Parameter	**UNIT**	**Groundwater**				**National standards**		
		BH1	**BH2**	**BH3**	**BH4**	**WHO**	**SANS**	**DWAF**
pH		8,00	8,00	8,00	8,00	6,5−8,5	5,0−9,7	6,5−8,4
EC	mS/m	95,7	66,60	77,7	44,8	150	≤170	≤40
TDS	mg/ℓ	482,00	332,00	386	300	−	≤120	−
DO	mg/ℓ	3,27	5,04	6,22	3,01	−	−	−
Salinity	mg/ℓ	0,48	0,33	0,38	0,29	−	−	−
TH	mg/ℓ	391,04	252,02	317,70	244,59	−	−	−
Alkalinity	mg/ℓ	344,57	301,63	301,50	295,64	−	−	−

Parameter	UNIT	Groundwater				National standards		
		BH1	BH2	BH3	BH4	WHO	SANS	DWAF
Ca	mg/ℓ	94,55	50,70	76,49	44,95	75	–	–
Mg	mg/ℓ	37,63	30,45	30,77	32,15	30	–	–
Na	mg/ℓ	98,52	89,11	73,91	46,84	200	–	70
K	mg/ℓ	3,35	7,32	3,36	2,34	300	≤200	–
Cl	mg/ℓ	140,49	92,06	81,68	9,32	300	≤300	100
NO_3^-	mg/ℓ	0,19	0,16	0,835	0,16	50	≤11	–
NO_2^-	mg/ℓ	0,02	0,01	0,011	0,01	3	≤0.9	–
SO_4	mg/ℓ	66,56	23,79	39,03	16,89	500	≤500	–
Al	mg/ℓ	0,06	0,05	0,05	0,04	0,1	≤300	5
As	mg/ℓ	0,00	0,00	0,00	0,00	0,01	≤10	0,1
Sb	mg/ℓ	0,00	0,00	0,00	0,00	0,02	≤20	–
Ba	mg/ℓ	0,05	0,02	0,033	0,018	1,3	≤700	–
B	mg/ℓ	0,04	0,05	0,04	0,04	2,4	≤ 2400	0,5
Cd	mg/ℓ	0,00	0,00	0,00	0,00	0,003	≤3	0,01
Co	mg/ℓ	0	0,00	0,00	0,00	–	–	0,05
Cr	mg/ℓ	0,01	0,00	0,01	0,01	0,05	≤50	0,1
Cu	mg/ℓ	0,00	0,00	0,00	0,07	2	≤2000	0,2
Fe	mg/ℓ	0,01	0,06	0,00	0,02	–	–	5
Mn	mg/ℓ	0,66	0,03	0,00	0,04	0,08	≤2000	0,02
Ni	mg/ℓ	0,00	0,00	0,00	0,00	0,07	≤400	0,2
Mo	mg/ℓ	0,06	0,00	0,01	0,01	–	≤70	0,01
Pb	mg/ℓ	0,00	0,00	0,00	0,00	0,01	≤10	0,2
Se	mg/ℓ	0,00	0,00	0,00	0,00	0,04	≤40	0,02
U	mg/ℓ	0,05	0,01	0,06	0,01	0,03	≤30	0,01
V	mg/ℓ	0,00	0,00	0,04	0,00	–		0,1
Zn	mg/ℓ	0,01	0,01	0,01	0,00	3	≤5	1
F	mg/ℓ	0,25	0,21	0,22	0,33	1,5	≤1.5	2
GOD		0,08	0,08	0,08	0,07			
RTt		50,00	60,00	50,00	52,00			

4.7.4 Comparisons in validation of vulnerability models

The validation methods are very useful tools in assessing the groundwater vulnerability in relation to groundwater results for a specific area. This gives decision-makers an outlook of the areas that are more susceptible to pollution in relation to others and where to conduct development. For the present study, the use of GOD and RTt were selected and used in conjunction with each other so as to present more accurate and reliable vulnerability indices for the study area. The methods did not differ extremely regarding their overall calculated vulnerability classes, although their parameters varied greatly where the GOD method only uses three parameters while RTt uses more than three.

The two methodologies produced distinct vulnerability indexes that both encompassed the whole study area. The GOD technique indicated very low vulnerability, but RTt exhibited medium vulnerability indices for all boreholes. From the correlation plots in Figure 4.8, the GOD plot showed a positive correlation to more parameters such as average trace elements, including both Pb and Cd, while the RTt method expressed a negative correlation with most of the parameters, and this was noted by an increasing concentration of parameters with a subsequent decrease in the vulnerability index. The linear regression plots shed more light into this observation. In all the linear regression plots in Figures 4.11 the GOD index showed a positive correlation where increasing the GOD index corresponded with increasing concentrations of parameters; however, an increase in the RTt index was followed by a decrease in concentrations of parameters. On the linear regression plots, however, there existed discrepancies where the 'R' factors were not the same for plots of different vulnerabilities against parameters. For instance, for NO_3^- and average trace elements, except the EC plot, the R^2 for RTt was greater than that of GOD even though GOD depicted a positive correlation. The R^2 value for RTt versus NO_3^- was 0,1981, while GOD had 0,1248, while for trace elements RTt had 0,2312 and GOD computed 0,0078, According to the findings of Sener (2021), the GOD and DRASTIC methods were utilised, and validation results revealed linear regression values of 0,5426 for DRASTIC and 0,4621 for GOD, implying that the higher R2 for the DRASTIC map had more accuracy than the GOD method. In the current investigation, the RTt map was likewise more accurate than the GOD map, despite the favourable association. As a result, the vulnerability indices did not demonstrate significant differences between groups, which can be attributed to the current research area.

4.8 Soil quality

4.8.1 Physicochemical parameters

The soil's physicochemical properties reflect how the landfill has altered its quality. In contrast to water quality, only pH and trace elements were reported in this study for soil quality, owing to the fact that trace elements tend to remain in the environment in the soil due to their specific gravity, posing a threat to groundwater quality (Izah et al., 2017).

- **pH**

The results of pH for soil in the landfill are presented in Table 4.13. There were no standard limits provided by both the WHO (1999) and DEA (2008) for suitable soil pH. It was found that the mean pH of soil across the landfill was 6,5±0,2 which indicated an acidic to near neutral soil nature in both seasons. Choudhury et al. (2022) discovered similar soil pH results and argued for soil contamination by the acidic leachate generated by the landfill site, which can also be attributed to the current study. Despite the distance and depth of the soil extracted from the landfill for the current investigation, the pH of the soil did not vary significantly. This may be due to leachate transport from the landfill all the way into the vicinity of the landfill as a result of precipitation. Also, the clay soil found in the area may have allowed for free flow of leachate from the landfill to the vicinity due to less permeability, porosity and infiltration.

- **Trace metals**

The trace elements used for this investigation were compared to the standards established by the DEA (2008) and the WHO (Nyika et al., 2019). Table 4.13 shows the results for soil trace elements.

Table 4.8 Mean geochemistry of soil around the landfill site in comparison with national standards

Parameters	**Soil samples**					**National Standards**	
	Unit	**Min**	**Max**	**Mean**	**±SD**	**DEA (2008)**	**WHO (1996)**
pH		6,36	6,64	6,50	0,19	–	–
V	mg/ℓ	166,00	166,00	166,00	0,00	150	2
Cr	mg/ℓ	179,00	181,00	180,00	1,41	6,5	100
Co	mg/ℓ	35,00	38,00	36,50	2,12	300	50
Ni	mg/ℓ	59,00	63,00	61,00	2,83	91	35
Cu	mg/ℓ	49,00	61,00	55,00	8,48	16	36
Zn	mg/ℓ	84,00	85,00	84,50	0,71	240	50
As	mg/ℓ	10,00	11,00	10,50	0,71	5,8	20
Mo	mg/ℓ	<1,00	<1,00	<1,00	0	–	–
Ba	mg/ℓ	613,00	697,00	655,00	59,397	–	–
Pb	mg/ℓ	13,00	14,00	13,50	0,70711	20	100
U	mg/ℓ	2,00	2,00	2,00	0	–	–

The measured trace elements from the landfill site were V, Cr, Co, Ni, Cu, Zn, As, Mo, Ba, Pb, and U, which had the mean concentrations of 166±0,1; 180±1,4; 36,5±2,1; 61±2,8; 55±8,4; 84,5±0,7; 10,5±0,7; <1±0; 655±59,3; 13,5±0,7; and 2±0, respectively. The soil component in the landfill site showed contamination due to elevated concentrations of V and Cr that exceeded both the DEA (2008) and the WHO thresholds (Nyika et al. 2019), while Cu and Zn only exceeded the WHO (Nyika et al., 2019), but fell below the DEA (2008) threshold (Figure 4.12). As only exceeding the DEA (2008) but falling short of the WHO criteria, Ba had the highest concentration of all trace elements in the garbage, despite the lack of any national or international limits in place. The current study's results were significantly higher than those reported by Odhiambo et al. (2015) and Ololade et al. (2019). Nyika et al. (2019) reported similar findings in their research of heavy metal pollution and mobility near a landfill in South Africa. Not all trace elements resulted in soil contamination, for instance, Mo, Pb, U, and Co all fell below all the standards for soil quality. The trace element concentration at the landfill site followed the trend Ba>Cr>V>Zn>Ni>Cu>Co>Pb>As>Mo.

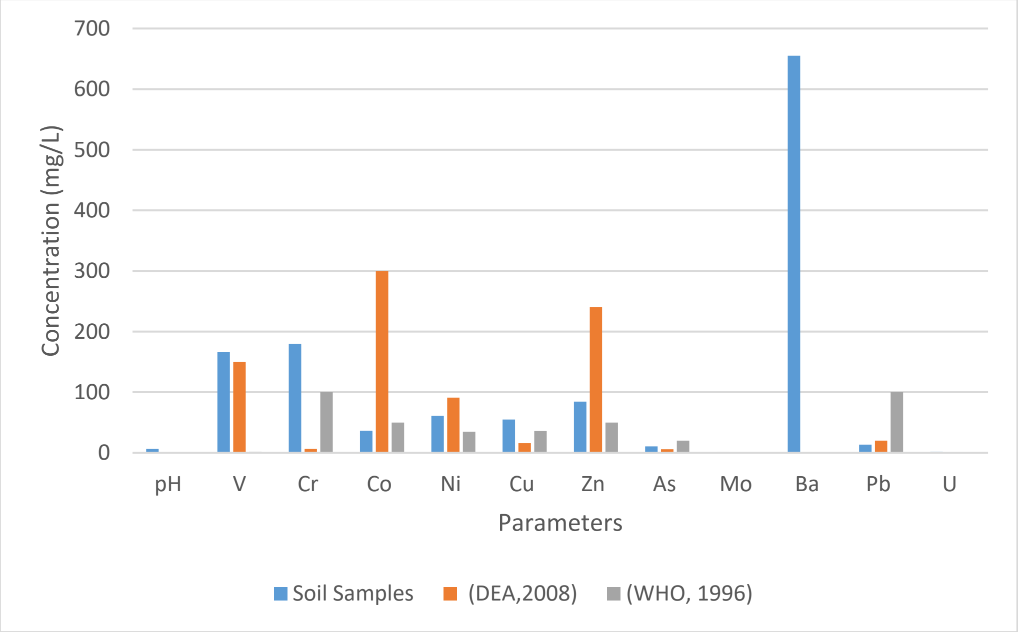


Figure 4.4 Soil quality parameter plots versus national standards

The presence of V, Ni, Cu, Zn, and As in soil could constitute a significant environmental danger due to leachate leakage into groundwater. A review conducted by Igboama et al. (2020) demonstrated that the existence of large quantities of these metals can be obtained from both natural and anthropogenic activities, such as geological features of the host environment that include topsoil and rocks as well as wastes that contain paints, batteries, pipes, and other metallic substances that find their way into landfill sites. Ba is sourced from the pigment industry and paints, and is often applied in ceramics, glass, detergents for motor oil, pottery, and glassware (Bawwab et al., 2022).

The Rustenburg layered suite of South Africa's Bushveld igneous complex naturally contains trace elements such as Cr, Cu, Ni, As, V and Zn. As a result, the weathering of gabbronorite rock in the bushveld sequence produces acid soils that may cause heavy metal enrichment. Consequently, the high amounts of these dangerous trace elements may have originated from solid waste dumping at the landfill site.

Soil pH is a driving force behind trace element mobility and accumulation; therefore, it can be used to assess heavy metal and nutrient availability in connection to toxicity and soil pollution (Sintorini et al., 2021). The alkalinity of pH is particularly important in controlling trace element accumulation at garbage dump sites since greater pH results in higher accumulation in the soil and vice versa. Several studies have observed mean alkaline pH values in soils at dump sites ranging from 8,103 to 9,80 (Odhiambo et al., 2015; Odom et al., 2021; Sintorini et al., 2021). Alkaline pH values were linked with elevated concentrations of trace elements that surpassed the standard limits or control samples due to low mobility of trace elements in the soil media, which leads them to settle or accumulate and increase in concentrations. However, the present study showed low acid pH values (less than a pH of 7) to be associated with very high concentrations of trace elements which contrasts with the phenomenon of high pH correlating with low mobility and accumulation of trace elements. Under acid conditions where acid soils contain high trace element concentrations, the processes of solubility and low adsorption can take place. Acidic soils tend to have high hydrogen ions (H+) that react with trace elements and replace their ions (Cu, Zn, Fe, and Mn) at the soil exchange sites and thus disseminating them into the soil solution where they become accessible to plants, thus, increasing trace element concentrations. Also, the nature of the lithology has a great impact on the trace elment concentration. Despite anthropogenic contamination, rocks and soils containing minerals such as montmorillonite and kaolinite aluminosilicate minerals tend to contain significant trace elements (Lei et al., 2020). The clay soils in the current study also contained significant aluminosilicate minerals such as smectite, plagioclase feldspar, k−feldspar, and quarts, which may be the reason for the high trace element enrichment due to adsorption under acid conditions (Table 4.14 and Figure 4.13).

Table 4.9 Mineralogical composition of the soil in the landfill site

Mineralogy	Mean Concentration (%)
Quarts	38,00
Plagioclase	15,50
K−feldspar/Rutile	3,50
Calcite	4,00
Kaolinite	5,50
Smectite	18,00
Mica	4,00
Clinopyroxene	4,00
Goethite	4,00

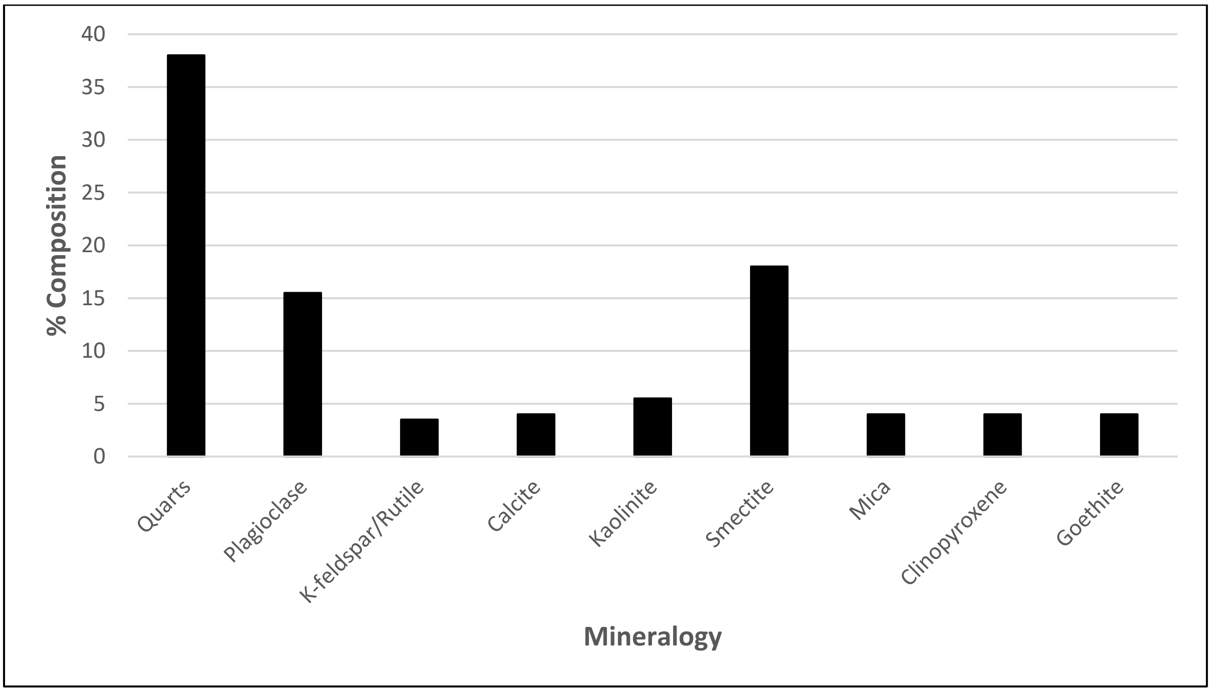


Figure 4.5 Mineralogical composition of soil around landfill site

4.8.2 Soil quality indices

Pollution indices in soils are an excellent technique to assess contamination. Soil pollution indices are used as a tool and pathfinder for the geochemical assessment of the state of the soil (Mazurek et al., 2017). These indices afford the chance to evaluate both environmental and soil degradation risks (Kowalska et al., 2018). Furthermore, the indices assist in deciphering whether the source of pollution of the soil is due to anthropogenic or natural processes. For the purposes of this study, both CF and the geoaccumulation and the Nemerow comprehensive indices have been used as they are also commonly used in conjunction in various literature to provide a more comprehensive and accurate interpretation as both indices have their strengths and weaknesses. Similar to water quality indices, soil quality indices also convert complex geochemical data into a simple and easy manner that can be comprehended.

- **Contamination factor**

Table 4.15 shows the computed results for the CF, as well as the classification based on the degree of pollution. The computed CF findings revealed that fewer than half of the parameters poisoned the soil, while the remainder exhibited no pollution at all. The CF parameters Cr and Cu showed soil contamination. Cr had the highest mean value of 27,69 which was categorised as very high pollution in both seasons, while Cu recorded a mean value of 3,45, which was also classified as significantly high contamination in both seasons (Figure 4.14). Conversely, other trace elements contributed to low contamination, with the following mean values Co (0,12), Ni (0,66), Zn (0,35), and Pb (0,67), while V and As had moderate contamination with mean values of 1,11 and 1,805, respectively. The study by Mavakala et al. (2022) on the dumpsites in developing countries also showed a high CF for Cr that exceeded the value of 1, whereas Fonge et al. (2017) recorded a high CF for Cu in a landfill in Mount Cameroon that was calculated at >6. The presence of these metals is ascribed to leachate contamination from landfills and agricultural sources, which frequently contain elevated levels of trace elements such as Cu and Cr. These trace elements are employed in manufacturing, and their by-products are commonly used in households, insecticides, fertilisers, herbicides, and fungicides, and compost production at landfill sites has the potential to damage soils (Fone et al., 2017; Luc et al., 2020).

Table 4.10 Mean values of the contamination factor and their classification

Trace elements	Contamination factor values for soil samples			
	Minimum	Maximum	Mean	Classification
V	1,11	1,11	1,11	Moderate contamination
Cr	27,54	27,85	27,69	Very high contamination
Co	0,11	0,13	0,12	Low contamination
Ni	0,64	0,69	0,66	Low contamination
Cu	3,10	3,81	3,45	Considerably high contamination
Zn	0,35	0,35	0,35	Low contamination

Trace elements	Contamination factor values for soil samples			
	Minimum	Maximum	Mean	Classification
As	1,72	1,89	1,80	Moderate contamination
Mo	–	–	–	–
Ba	–	–	–	–
Pb	0,65	0,70	0,67	Low contamination
U	–	–	–	–

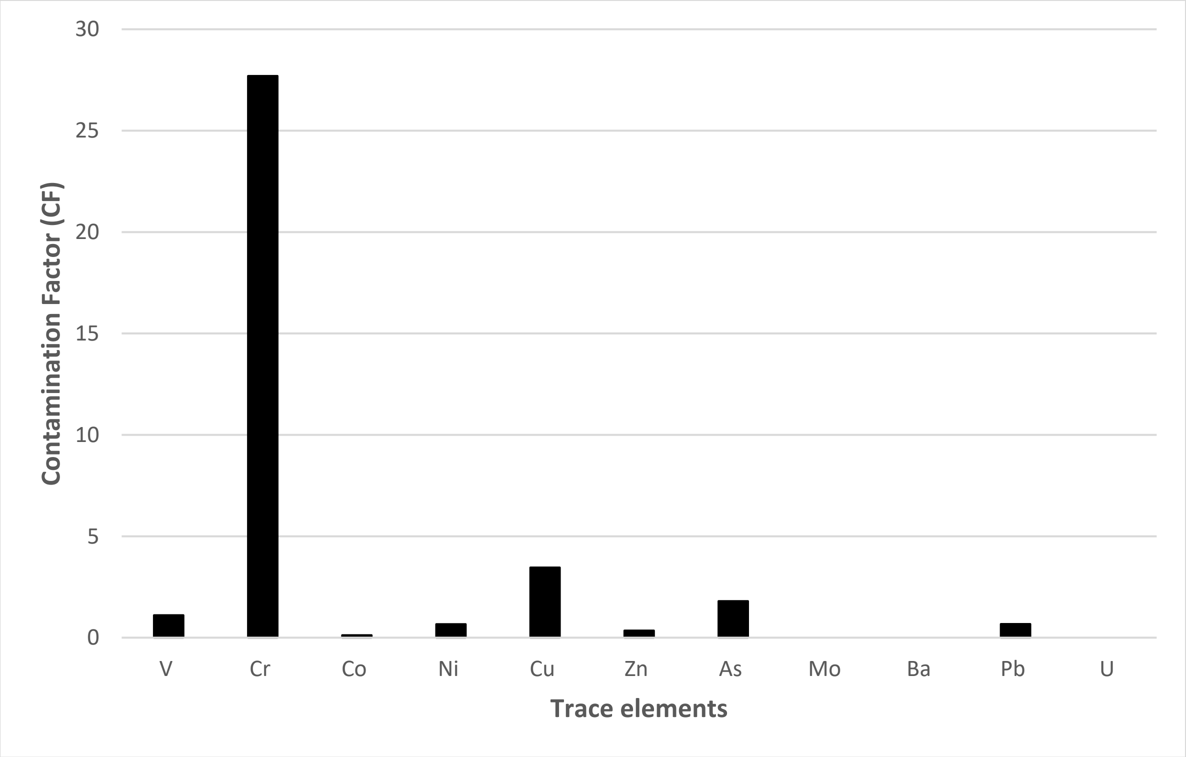


Figure 4.6 Contamination factor versus trace elements in soil

The CF for As and Zn in the research by Ahmad et al. (2021) and Fonge et al. (2017) recorded moderate contamination with values of 3,43 and 2,25, respectively, which agrees with the results of this study. Pb also contributed a moderate contamination at 2,97, which contrasted with the low contamination results of this study. Several studies at landfill sites reported high to very high contamination contributed by Co, Ni, Zn, and Pb with the values greater than 6, which contrasts with the low contamination contribution in the present study (Ahmad et al., 2021; Mavakala et al., 2022). V recorded moderate contamination, which may be attributed to an anthropogenic source introduced into the landfill by vanadium-containing batteries that come into contact with soils (Gustaffon, 2019). The low CF in the landfill may signify less contribution of the leachate in relation to these trace elements. As a result, only Cr and

Cu contributed very high contamination of the soil around the landfill for contamination factor.

- **Geoaccumulation index**

Table 4.16 reports the results of geoaccumulation index as represented in Figure 4.15. The geoaccumulation index, along with CF, is one of the most extensively used methodologies for assessing soil quality indices by researchers throughout the world. The table shows that only Cr and Cu were the contaminating trace elements at the dump site. The categorisation of the geoaccumulation index is divided into five classes: very strong (>5), strong to very strong (>3−4), moderate (>1−2), almost uncontaminated to moderate (>0−1), and practically uncontaminated (<0). The mean Cr and Cu levels were 4,2 and 1,16, respectively, indicating significant to very strong and moderate contamination. This shows that the landfill site has an anthropogenic influence as a result of the disposal of mixed electronic, household, and commercial trash (Fone et al., 2017). The results of the geoaccumulation index were validated by the geoaccumulation index technique, which also identified similar contaminants. The remaining elements (V, Co, Ni, Zn, As, and Pb) were categorised as practically uncontaminated, with negative mean values smaller than 0 in both seasons. Essien et al. (2022) and Mahakala et al. (2022) investigated the soil quality around municipal landfills and found uncontaminated and negative results in soils contaminated with Cr, which contradicted the findings of the current study. Ahmad et al. (2021) found that Cr (1,14) and Cu (1,0) made a moderate contribution to anthropogenic activities. Fone et al. (2017) also showed a moderate contribution by Cu (2,49), which accords with the results of the current study; however, Jafaru et al. (2015) reported opposing results of high pollution by Cu at a dump site in Accra, Ghana. Comparing the geoaccumulation index results to the soil quality values in chapter 4 revealed Cr and Cu contamination. Cr exceeded both the DEA (2008) and WHO limits (Nyika et al., 2019), whereas Cu only exceeded the WHO standards (Nyika et al., 2019) throughout both the wet and dry seasons. The geoaccumulation index correlates to water quality data for both Cr and Cu, but differs with As and Zn, which surpassed the WHO norm, signifying contamination but were below the DEA (2008) threshold limit. V also exceeded all the standard limits for WHO and DEA (2008) but recorded no contamination contribution for the geoaccumulation index. It also agreed with low concentrations of Pb and Co, which fell below all the standard limits prescribed for soil usage in both seasons. Trace elements such as Mo, U, and Ba had no standards for usage, and, therefore, could not be used for Igeo. From the comparisons of both the geoaccumulation index and soil quality, it can be seen that the geoaccumulation index provided accuracy in most of the contributing single parameters such as the CF method that affected the soil in landfill site, even though it could not account for discrepencies in contamination by the parameters themselves.

Table 4.11 Geoaccumulation index values and classification for soil in the landfill

Trace elements	Geoaccumulation Index values for soil samples			
	Minimum	**Maximum**	**Mean**	**Classification**
V	−0,44	−0,44	−0,44	Not polluted
Cr	4,19	4,21	4,20	Strong to very strong
Co	−3,56	−3,68	−3,62	Practically uncontaminated
Ni	−1,11	−1,21	−1,16	Practically uncontaminated
Cu	1,03	1,30	1,16	Moderate
Zn	−2,08	−2,10	−2,09	Practically uncontaminated
As	0,20	0,34	0,27	Practically uncontaminated
Mo	−	−	−	−
Ba	−	−	−	−
Pb	−1,10	−1,20	−1,15	Practically uncontaminated
U	−	−	−	−

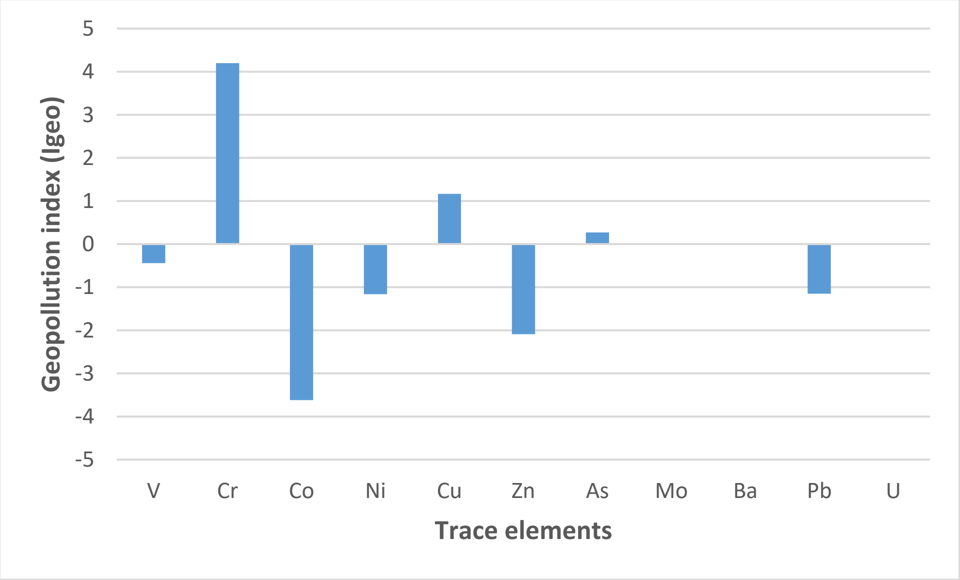


Figure 4.7 Geoaccumulation index versus trace elements for soil

- **Nemerow comprehensive index**

In contrast to the CF and geoaccumulation indices, the NCI considers the overall pollution caused by all of the components included, rather than the contribution of any single parameter. Table 4.17 displays the results of the NCI, which were computed using all of the trace elements that were analysed at the dump site. Table 4.18 shows the classification of the NCI values: no pollution (≤0,7), slight pollution (0,7−1), moderate pollution (1−2), severe pollution (2−3), and extremely severe pollution (>3). The NCI data from Table 4.17 indicated a mean value of 470,215 for the soil around the waste site, indicating extremely serious contamination. This shows the overall trace element contamination effect of the landfill site on the soil in both seasons, thus, rendering it unfit for usage for all land uses. A case study conducted by Liu et al. (2015) using various analysis methods for environmental quality assessments reported 37,5% of the soil samples to be extremely contaminated due to anthropogenic activities with NCI values of 8,88, 32,63, and 198,03. These results corresponded with the results of the current study where they reported a decline in the soil qualitities of the individual samples. Significantly lower NCI values than that of the present study were found by Zhou et al. (2022) at the Shannan landfill site, China. They reported mean values ranging from 2 to 3, indicating that the dump site supplied moderate pollution to the soil surrounding the landfill, which was linked to soil formation processes as well as the impact of human activities. In the current study, anthropogenic influence of the landfill site may have been the primary driver of such severe contamination. The NCI also corresponded with the soil quality data reported above, indicating that trace elements such as As, Ba, Cr, Cu, V and Zn contaminated the soil around the dump site, surpassing DEA (2008) and WHO guidelines (Nyika et al., 2019).

It can be noted that soil quality index methods complement one another as they all computed similar results regarding the state of contamination in the landfill site. All the methods reported contamination of soil as a result of the lanfill site. Authors such as Kowalska et al. (2018) have reviewed various kinds of soil quality index methods regarding their weaknesses, strengths, and applications. And when they are used in conjunction with one another to analyse for soil quality index, they often deliver more accurate results than when they are used individually. For the current investigation, both individual and complicated approaches were applied. The CF and the geoaccumulation index rely on individual trace elements as contaminating variables, but the NCI defines contamination holistically by utilising the sum of all trace elements in a soil medium. The methods, therefore, performed well as they all corresponded to the soil quality results. This gives decision-makers, researchers, and stakeholders a bird’s eyeview that simplifies the state of the soil quality for better decision-making.

Table 4.12 The value for Nemerow comprehensive index of soil in the landfill

Nemerow comprehensive index for soil samples			
Min	**Max**	**Mean**	**Classification**
440,43	500	470,215	Extremely severe contamination

Table 4.13 Pollution index methods and their classifications

Index method	Range	Classification	References
CF	<1	Low contamination	Devanesan et al. (2017), Jafaru et al. (2015)
	>1−3	Moderate contamination	
	>3−6	Considerably high contamination	
	>6	Very high contamination	
Geoaccumulation index	>5	Very strong	Jafaru et al. (2015)
	>4−5	Strong to very strong	
	>3−4	Strong	
	>2−3	Moderate to strong	
	>1−2	Moderate	
	>0−1	Practically uncontaminated to moderate	
	>0	Practically uncontaminated	
NCI	≤0,7	No pollution	Shan et al. (2022)
	0,7−1	Slight pollution	
	1−2	Moderate pollution	
	2−3	Severe pollution	
	>3	Extremely severe pollution	

Chapter 5

Conclusions and Recommendations

5.1 Introduction

This chapter draws findings from the study and makes appropriate recommendations based on some of the observations made. This chapter also contains information about the limits found during the course of the study.

5.2 Conclusions

The study's conclusions were based on the results of the water and soil physicochemical characteristics, geochemistry, water and soil quality indices, and groundwater vulnerability. The following conclusions were drawn in response to the objectives of the study.

Objective 1:
To assess groundwater, surface water, and soil quality using physicochemical parameters in comparison with the leachate chemistry to decipher whether there is contamination.

The leachate from the Botshabelo landfill site was exceedingly hazardous. The leachate chemistry revealed higher amounts of K, Na, and Cl ions compared to groundwater and surface water. The prevalence of Na, K, and Cl in the leachate was linked to the high salt content of the waste coming from the general waste types from domestic settings. Also, the leachate had high alkalinity and TH that exceeded the groundwater and surface water, which was asserted to have come from the combustion of agricultural waste and wood. All the trace element concentrations in the leachate exceeded those in groundwater and surface water. However, only Mn and Mo in one groundwater sample were higher than in the leachate. This may have been attributed to element transport by the infiltrating leachate, coupled with a closer distance to the waste heap. Additionally, the leachate had very low concentrations in comparison to the clay soil surrounding the landfill site, which could be attributed to its tendency to combine with trace elements with each episode of trace element deposition by leachate, as well as their ability to remain in the environment for long periods of time.

Objective 2

To determine groundwater, surface water, and soil quality using physicochemical parameters in comparison with the local and global standards. This is to determine how the landfill affected the water and soil qualities over two seasons.

Over two seasons, the mean pH of the groundwater and surface water at the dump site was alkaline, falling below the WHO, SANS, and DWAF limitations. The pH values in the dump indicated a mature waste stabilisation phase associated with an old landfill. The mean EC for groundwater surpassed the DWAF limits for both seasons, with BH1 having the greatest concentration, which were within the WHO and SANS acceptable levels; however, surface water had a mean EC value that was lower than all of the stipulated standards. The mean TDS concentrations in all boreholes and surface water samples were lower than the WHO and SANS drinking quality criteria in both seasons; however, there were no DWAF standards specified, indicating no contamination problem. The DO concentrations in all water samples were very low, indicating the existence of higher organic matter as a result of the landfill, which consumes available oxygen and creates both depletion and shortage. TH had no water quality standards but was rated as hard for groundwater and fairly hard for surface water. This was linked to the geology of the area, which consisted of sedimentary rocks 3like siltstones interspersed with layers of clay and shale. Alkalinity levels in both groundwater and surface water were higher than any other metric measured. High alkalinity at the dump site was associated with alkaline pH in both seasons, which may have caused the water to have an unpleasant taste, thus, deeming it unfit for drinking.

Furthermore, the mean concentrations of Na, K, Ca, SO_4, Cl−, NO_3^-, and NO_2^- for both groundwater and surface water in both seasons were lower than national standards, indicating a low pollution concern from the landfill and local agricultural farms. However, Mg levels in both seasons were above the WHO guidelines, indicating pollution. The Mg enrichment may have been caused by local geology, specifically the intrusion of dolerite dykes. In addition, the degradation and dissolving of inorganic solid waste products from landfills, such as tiles, plaster, and concrete, tend to increase cations. The Stiff graphs verified the results of the water quality measurements for main ions and alkalinity concentrations. The mean trace element concentrations of groundwater and surface water over the course of two seasons showed no contamination to groundwater because they were all below the national criteria, except for Mn. Excess mean concentrations of Mn were reported to exceed the WHO limit but were below the SANS limit. This may be resulted by the anthropogenic activities and Mn containing leachate from the landfill site due the dumping of wastes such as batteries and electronic components, coupled with element substitution of iron-bearing minerals such as magnetite or rhodochrosite. Also, the low groundwater concentrations may have been due to the confinement of the aquifer to the impermeable clay and mudstone layers dominant in the landfill site acting as a protective cap against infiltrating pollutants. Over two seasons, there were no notable variations in the concentrations of physicochemical parameters at the Botshabelo non-engineered landfill.

Objective 3

To assess the geochemistry of groundwater, surface water, and soil over two seasons to understand their evolution.

The study's findings revealed that both surface water and soil were classified and represented the $CaMgHCO_3$ type of water regarding samples BH4, SWu, and SWd in both seasons, indicating that they contained very little mineralisation in terms of main ions, indicating that the water was fresh and recently refilled with groundwater. CaMgCl type of water was indicative of BH1, BH2, and BH3 on the Piper plot. The cation content of the water around the Botshabelo landfill revealed no dominant type for samples BH3, BH4, and SW_u, while a Mg type of water was shown by the samples BH2 and SW_d. Moreover, a mixed zone in the Piper plot was shown to contain samples BH1, BH2, and BH3 concerning the anion content. The HCO_3^- water type was shown by the BH4, SW_u, SW_d samples. Organic matter in the aquifer causes carbon dioxide to be converted to bicarbonate under oxidising circumstances. Also, in addition to natural processes, human activities are significant external variables that can change the hydrochemical composition of water. As a result, water−rock interaction, evaporation, carbonate weathering, and anthropogenic sources all have a significant impact on the chemistry of the water bodies studied.

The pH of the soil surrounding the dump site was found to be acidic, indicating an acidic environment. This could have been caused by acidic leachate from the landfill that was distributed throughout the soil. The study’s findings revealed a trend of heavy soil contamination by trace elements Ba>Cr>V>Zn>Ni>Cu>Co>Pb>As>Mo. Predominatly contaminating elements were V and Cr, which exceeded the national standards of the DEA and WHO. Cu and Zn only exceeded the WHO standards, but fell below that of the DEA. Ba had the highest concentrations of all the trace elements but had no national standard limits. As also exceeded the DEA standards, but fell below that of the WHO. Mo, Pb, U, and Co had concentrations that all fell below the national standards, signalling no contamination effect. The heavy contamination in the soil can be attributed to the impact of the landfill due to dumping of wastes such as paints, pigments, batteries, and electronic appliances which are applied in different areas, including ceramics, glass, detergents, pottery, and glassware. The mineralogical composition of soil in the Botshabelo non-engineered landfill is typical of aluminosilicate such as smectite, plagioclase−feldspar, k−feldsapr, and quarts that indicate that the soil is predominantly clay. The mineralogy also supports the heavy trace element enrichment of the clay due to their high affinity and adsorption capacity to form complexes with trace elements.

Objective 4

To simplify the water and soil quality results by the application of groundwater, surface water, and soil quality indices.

The results of NPI, employed in connection to the WHO and DWAF criteria for drinking and irrigation, revealed no pollution, with mean values less than 1 for all key elements except Mg, which surpassed the limit of 1, Except for Mg, which polluted somewhat in comparison to the WHO limit, all of the key elements caused no pollution. However, no contamination was observed by main elements for the DWAF criteria. Slight pollution may be caused by dumping municipal solid trash in the Botshabelo non-engineered landfill. During the wet season, EC, Mn, and U polluted groundwater at the landfill site to varying degrees. The remaining trace components showed no contamination for both WHO and DWAF requirements in all seasons. Based on irrigation water quality Mo and U posed serious pollution to groundwater in the wet season, while B, Na, and manganese all contributed slight and serious pollution Human activities such as landfills, industrial discharges, mining, leaching, emissions, fertilisers, and coal combustion are predominantly primary sources of both Mn and U. The surface water around the landfill was classified as moderately polluted by the parameters by manganese only in the wet season for irrigation. The WAWQI's complete index rated groundwater at the dump site as good to exceptional for drinking purposes, but irrigation water varied from poor to unfit for drinking reasons and required sufficient treatment before use in both seasons. All of the surface water samples were rated excellent for drinking; however, they ranged from excellent to good for irrigation. The NSFWQI produced data that differed from the WAWQI in terms of overall WQI categorisation near the dump site. The NSFWQI evaluated the groundwater as very poor for drinking in all seasons, which is consistent with the NPI readings. Surface water quality ranged from fair to bad for drinking in all seasons. All three WQI methodologies produced a more accurate categorisation, indicating a reduction in groundwater and surface water quality as a result of landfilling.

The contamination factor data revealed individual trace components that caused pollution at the dump site. V, As, Cu, and Cr in soil around the landfill contribute moderate, significant, and very high pollution, whereas the rest of the trace elements provide low contamination. The leachate contamination, as well as agricultural sources, were linked with Cu and Cr, which are often applied in households, pesticides, fertilisers, herbicides, and fungicides. V was associated with vanadium-containing batteries often dumped in municipal solid waste dumping sites. Similar results were reported by the geoaccumulation index for Cr and Cu. which both contributed strong to very strong contamination. This contamination was caused by the landfill site's anthropogenic influence by the disposal of mixed electronic, household, and commercial garbage. Cu caused moderate pollution, but the other trace metals did not pollute the dump site. The results of Nemerow's thorough index revealed that the overall pollution in the dump was badly contaminated, rendering it unfit and unsuitable for any land use. This demonstrated the overall degradation in soil quality around the Botshabelo non-engineered landfill site as a result of dumping municipal trash over time.

The use of the CF, the geoaccumulation index, and NCI in conjunction with one another, revealed better and more accurate results as they signalled corresponding results.

Objective 5

To determine the susceptibility of groundwater to pollution by the application of rainfall travel time (RTt) and the groundwater occurrence or aquifer type, overall lithology of aquifer, and depth of groundwater (GOD) vulnerability methods, and to delineate vulnerable areas for future decision-making.

The RTt and GOD methods were used to calculate groundwater vulnerability in the research area because they were more appropriate for the land-use type. The study revealed a very low vulnerability class that covered 100% of the area in the landfill site according to the GOD index, which was attributed to the aquifer confinement to impermeable mudstone strata and the clay cover on the land surface acting as a protective cap against emerging contaminants. On the other hand, the RTt index calculated a moderate vulnerability that similarly covered 100% of the entire landfill site. The moderate vulnerability was ascribed to the low rainfall since the overall climate is of a semi-arid nature, coupled with low porosity, moderate slope as well as the low flow hydraulic conductivity of clay which were all the contributing factors.

The vulnerability methods were validated on their performance and to gauge contamination by using selected physicochemical parameters such as NO_3^-, EC, and trace elements. In both methods, the GOD index showed a more positive correlation as well as a positive linear regression whereby very low concentrations of water quality parameters corresponded with the low vulnerability class of groundwater. The RTt method, however, revealed a negative correlation underpinned by increasing vulnerability causing a decrease in concentrations of parameters. Negative linear correlations were depicted by the RTt model. Discrepancies were noticed on the values of the R-factor which showed the RTt method having higher values than those of the GOD method which was concluded to have high accuracy in its maps than the GOD model despite negative correlation results it revealed.

5.3 Recommendations

In South Africa, only a few studies focused on the influence of non-engineered landfill sites in water resources and soil, and how they tend to deteriorate over time. This study, therefore, adds to the body of research that has already been amassed by various studies in South Africa and it provides more insight into information and data towards related studies. Although the time period of the study was over a short period of time, the following recommendations can be made on the study area:

- The landfill site should have leachate collection systems built around the landfill site to collect the leachate and prevent it from overflowing into water resources and soil medium. This will curb contamination to be dispersed during episodes of high rainfall during summer seasons.
- Since the clay cover on the surface of the landfill acts as a protective shield against downward infiltration of contaminants on the surface, there more research should be conducted that include soil borings, core samplings and geophysical methods in order to determine if the contaminants could potentially reach the aquifer since groundwater occurs at shallow levels around the landfill site. Also, the insertion of liners can be done in the study area so as to shield the leachate from percolating downwards.
- The opened boreholes that are both inside and outside the landfill should be closed by means of a borehole cap to prevent the influx of pollutants that come from the landfill site itself.
- Experienced geohydrologists should monitor surface water and groundwater at the landfill site to assess potential risks from emerging contaminants. This will provide valuable data for future research. Also, there should be a detailed groundwater vulnerability study using various groundwater vulnerability methods.
- The opened boreholes that are both inside and outside the landfill should be closed by means of a borehole cap to prevent the influx of pollutants that come from the landfill site itself. More boreholes should be drilled around the landfill site for more groundwater data acquisition.

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Appendix A

Supplementary information to Chapter 4

Water Quality Results

	Wet season						Dry season								
	BH1	**BH2**	**BH3**	**BH4**	**SW (d)**	**SW (u)**	**BH1**	**BH2**	**BH3**	**BH4**	**SW (d)**	**SW (u)**	**WHO**	**SANS**	**DWAF**
Ca	91,39	51,61	75,89	46,13	30,45	30,2	97,70	49,79	77,09	43,76	28,03	28,47	75	–	–
Mg	36,84	25,59	30,88	32,7	12,44	14,2	38,42	35,32	30,66	31,59	13,51	13,62	30	–	–
Na	105,69	63,17	81,13	50,43	16,3	19,55	91,35	115,06	66,69	43,25	26,21	16,36	200	–	≤70
K	2,41	7,32	1,99	1,37	4,06	2,19	4,29	7,32	4,73	3,31	5,11	6,51	300	≤200	–
Cl	131,4	64,28	72,22	6,21	7,50	6,71	149,58	119,84	91,14	12,42	20,73	13,37	200−300	≤300	≤100
NO_3	<0,16	<0,16	0,65	<0,16	<0,16	<0,16	0,23	<0,16	1,02	<0,16	<0,16	<0,16	50	≤11	–
NO_2	0,02	0,014	0,012	0,007	0,012	<0,006	0,02	0,01	<0,01	0,01	0,01	<0,01	3	≤0,9	–
SO_4	67,02	18,63	40,97	19,43	5,24	4,70	66,11	28,95	37,09	14,35	12,91	7,40	500	≤500	–
Alkalinity	341,10	250,28	295,99	296,59	162,42	172,96	348,05	352,97	307,02	294,70	172,77	162,27	–	–	–
Ca hardness	228,20	128,88	169,51	115,19	76,04	75,49	243,96	124,33	192,49	109,47	69,98	71,09	–	–	–
Mg hardness	151,71	105,37	127,14	134,67	51,24	58,38	158,21	145,45	126,26	130,07	55,65	56,08	–	–	–
Total hardness	379,91	234,25	316,65	249,85	127,28	133,88	402,17	269,78	318,75	239,34	125,63	127,18	–	–	–
Al	0,099	0,080	0,084	0,062	0,058	0,049	0,02	0,01	0,01	0,01	0,13	0,02	<0,1	≤300	≤5
As	<0,00	<0,00	<0,00	<0,00	<0,00	<0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,01	≤10	≤0,1
Sb	<0,00	<0,00	<0,00	<0,00	<0,00	<0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,02	≤20	–
Ba	<0,09	0,041	0,066	0,036	0,053	0,035	0,00	0,00	0,00	0,00	0,00	0,00	1,3	≤700	–
B	0,018	0,043	0,016	0,031	0,017	0,020	0,06	0,05	0,06	0,06	0,04	0,04	2,4	≤ 2400	≤0,5
Cd	<0,00	<0,00	<0,00	<0,00	<0,00	<0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,003	≤3	≤0,01
Co	<0,00	<0,00	<0,00	<0,00	<0,00	<0,00	0,00	0,00	0,00	0,00	0,00	0,00	–	–	≤0,05
Cr	0,018	0,007	0,016	0,02	<0,00	<0,001	0,001	0,001	0,001	0,002	0,002	0,002	0,05	≤50	≤0,1
Cu	0,001	0,003	0,001	0,068	0,027	0,127	<0,01	<0,01	<0,01	<0,01	<0,01	<0,01	2	≤2000	≤0,2
Fe	0,008	0,071	0,003	0,024	0,050	0,102	0,01	0,05	<0,01	<0,01	0,10	0,01	–	–	≤5
Mn	0,857	0,035	0,001	0,068	0,027	0,127	0,46	0,03	0,003	0,004	0,005	0,01	0,08	≤2000	≤0,02
Ni	<0,00	<0,00	<0,00	<0,00	<0,00	<0,00	0,002	0,001	0,001	0,001	0,003	0,002	0,07	≤400	≤0,20
Mo	0,120	0,006	0,014	0,014	0,000	<0,00	0,012	0,002	0,003	0,001	0,002	0,001	–	≤70	≤0,01
Pb	<0,00	<0,00	<0,00	<0,00	<0,00	<0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,1	≤10	≤0,2
Se	<0,00	<0,00	<0,00	<0,00	<0,00	<0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,04	≤40	≤0,02
U	0,097	0,014	0,114	0,018	<0,00	<0,00	0,01	0,003	0,011	0,002	0,003	0,000	0,03	≤30	≤0,01

	Wet season						Dry season								
	BH1	**BH2**	**BH3**	**BH4**	**SW (d)**	**SW (u)**	**BH1**	**BH2**	**BH3**	**BH4**	**SW (d)**	**SW (u)**	**WHO**	**SANS**	**DWAF**
V	<0,00	<0,00	0,073	0,000	0,047	<0,00	0,001	0,000	0,009	0,003	0,007	0,001	–		≤0,10
Zn	0,005	0,004	0,008	0,005	0,004	0,004	0,011	0,010	0,010	0,001	0,001	0,001	3	≤5	≤1
F	0,26	0,26	0,25	0,38	0,23	0,21	0,23	0,16	0,18	0,28	0,48	0,20	1,5	≤1,5	≤2
PH	8,0	8,0	8,0	8,0	8,0	8,0	8,0	8,0	8,0	8,0	8,0	8,0	6,5–8,5	5–9,7	6,5–8,4
EC	95,7	66,6	77,7	44,8	27,9	32,7	93,6	67,5	77,4	44,8	27,9	33,5	150	≤170	≤40
TDS	482	332	384	300	140	163	482	336	384	305	139	167	500	≤1200	–
DO	3,27	5,04	6,22	3,01	3,17	3,72	3,30	5,0	6,22	3,00	3,2	3,70	–	–	–
Salinity	0,48	0,33	0,38	0,29	0,13	0,16	0,48	0,33	0,38	0,30	0,13	0,16	–	–	–

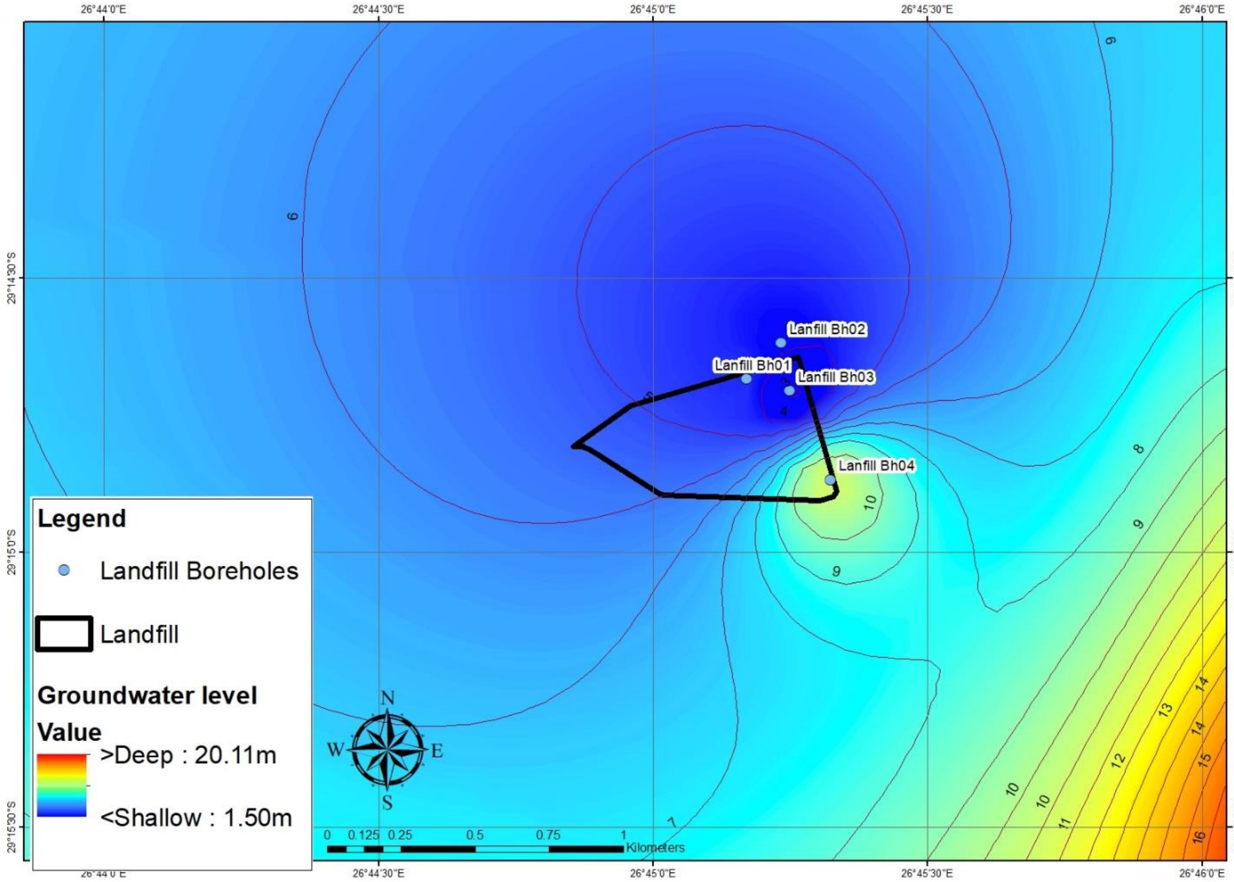
Note: Units of measurement for pH in pH units, EC in mS/m, TDS in PPM, Salinity in PSU, and the rest of the parameters in mg/ℓ.

Water quality indices results

WAWQI	*Wet season*		*Dry season*	
Sample point	**WHO**	**DWAF**	**WHO**	**DWAF**
BH1	43,234	922,581	15,060	308,900
BH2	6,139	63,533	1,737	62,210
BH3	23,987	276,850	2,378	271,100
BH4	8,344	107,591	0,956	9,628
SW(d)	2,534	35,289	3,211	11,670
SW(u)	4,939	68,847	0,804	7,523

NSFWQI **Sample point**	*Wet season*	*Dry season*
BH1	319,200	334,500
BH2	194,100	242,400
BH3	258,500	259,800
BH4	194,900	186,700
SW(d)	97,500	101,500
SW(u)	105,900	98,800

Groundwater level map



Appendix C

Turnitin Report

Appendix C

Language Editor's Report

Dorathea (Dora) du Plessis
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Confirmation of Editing and Proofreading

I confirm that I have done the proofreading and technical editing for the following master's dissertation:

Student:	Moeketsi Sesing
Student number:	222026439
Title:	The impact of the leachate from the Botshabelo non-engineered landfill on groundwater, surface water and soil quality
Degree:	Master of Health and Environmental Sciences
University:	Department of Life Sciences, Central University of Technology, Bloemfontein

Technical editing included the following: The dissertation was formatted using a custom MSWord template created specifically for this document. Technical editing also focused on the correct use of citations, style and formatting according to the Harvard referencing method. I made sure that all acronyms and abbreviations were used consistently throughout the text. I also double-checked the references in the chapters to ensure that the dates and spelling of author names used in the text matched those in the list of references. Wherever necessary, I used Google to find the correct information. I informed the student about any missing references that still needed to be added.

Language editing entailed maintaining consistency and accuracy in punctuation, spelling, sentence structure, and grammar. I tried to keep as much of the student's writing style as possible while making sure that the student's intended meaning was not altered during the editing process. All changes were marked with the Microsoft Word track changes feature. I also left comments on issues that the student needed to double-check or revise. The student was thus given the option of accepting or rejecting the recommended changes or suggestions.

I have 45 years of experience in typing, editing, and proofreading for postgraduate students at universities throughout South Africa and abroad. I gained further experience while working in various departments at the University of the Free State (UFS). I also helped to compile a document on technical layout and referencing methods. I presented a couple of guest lectures on referencing methods and technical layout issues to postgraduate students at the UFS. In the past couple of years, I have also proofread six books for publication, as well as several journal articles.

Disclaimer: The ultimate responsibility for accepting or rejecting the amendments and recommendations made using track changes rests with the student. The editor cannot be held responsible for any changes in terms of the format and style due to subsequent additions or deletions to the document, or any language issues that may have emerged as a result of subsequent amendments to the text.

Yours sincerely

Dorathea (Dora) du Plessis
Technical & Language Editor