ASSESSING THE IMPACT OF LANDFILL SITES ON SOIL AND WATER QUALITY IN NEIGHBOURING COMMUNITIES: A CASE STUDY OF GA-RANKUWA TOWNSHIP, GAUTENG PROVINCE OF SOUTH AFRICA

by

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DECLARATION

I declare that the above dissertation is my own work and that all the sources that I have used or quoted have been indicated and acknowledged by means of complete references.

I further declare that I submitted the dissertation to originality checking software and that it falls within the accepted requirements for originality.

I further declare that I have not previously submitted this work, or part of it, for examination at Unisa for another qualification or at any other higher education institution.

T.Matlakala

28 November 2025

SIGNATURE

DATE

DEDICATION

I would like to dedicate this dissertation to my late father, **TONE JOEL KGASWE.** I would not be who I am if it were not for his support and love. The Matlakala and Kgaswe family for building a fine, genuine person in me so that I can achieve anything beyond measure.

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I thank God for giving me the strength and wisdom to accomplish this dissertation in due time. They claim that your supervisor has a significant impact on your ability to succeed. Dr Radingoana Makgalake Pabalelo, my supervisor, undoubtedly did. I will be eternally grateful for her patient guidance, insightful comments and advice, and ability to be very forthright when I made a mistake. I thank my colleague, Mr Nthau, and my dear friend, Mr G Sikwane, for guiding and encouraging me to always aim high in life.

My further appreciation goes to the City of Tshwane-City Strategy and Organisational Performance department for approving my request to conduct my study in Ga-Rankuwa. I can never speak my gratefulness enough.

"And let us not grow weary of doing good, for in due season we will reap, if we do not give up."

~ Galatians 6:9

ABSTRACT

Globally, solid waste generation is increasing due to accelerated population growth, rapid urbanisation, and economic activities. Consequently, the inadequate, absent, and poor planning and implementation of waste management in Ga-Rankuwa results in more waste in landfills. Although landfills carry a huge amount of waste, unmanaged waste can cause an increase in greenhouse gases, a reduction in the aesthetic of the environment, and untreated leachate pollutes surrounding water and soils, especially when proper procedures and maintenance are not followed. Regardless of how much research and awareness are done on this waste management method, environmental contamination is still evident due to inadequate/improper monitoring.

The study aimed to assess the environmental impacts of the landfill site on soil and water quality in Ga-Rankuwa township. Soil samples were collected at the landfill perimeter in all 4 cardinal points and at 1 km and 2 km away from the 1st point, etcetera. Heavy metals, including Chromium, Mercury, and Lead, were assessed per sample for possible contamination by the landfill. Water samples were collected by directly dipping the containers in the nearby stream to collect water samples (upper river, mid river and lower river sections). The samples were taken to a laboratory for analysis, and further statistical tests and indices were used, and further analysis was conducted using XLSTAT. The water salinity and pollution indicators were assessed using pH, major cations (Na^+, K^+, Ca^{2+}), phosphorus and heavy metals in soil and river samples near a landfill.

Results showed significant variability in soil pH, with more acidic conditions closer to the landfill, likely due to leachate migration. Although heavy metal concentrations were elevated at certain sites, all values remained below WHO permissible limits and were classified as uncontaminated according to Müller's Geo-accumulation Index. River water showed slightly alkaline conditions, however, nutrient enrichment led to poor water quality. Trophic State Index (TSI) values for phosphorus exceeded 100 across all sites, classifying the river as hypereutrophic and identifying it as a pollution hotspot. While sodium values remained within acceptable limits for irrigation, microbial analysis revealed elevated amounts of E. coli and total coliforms downstream, indicating potential public health risks.

It is recommended that landfill operators adopt advanced engineering solutions, such as state-of-the-art leachate collection and treatment systems, to minimise the release of harmful contaminants into adjacent soil and water bodies. Furthermore, the integration of waste segregation at source and comprehensive recycling programs should be prioritised to reduce the volume of non-biodegradable and hazardous waste entering landfills.

Keywords: Environment, Ga-Rankuwa, heavy metals, landfill site, soil quality, waste management, water quality.

KAKARETŠO

Lefaseng ka bophara, tšweletšo ya ditlakala tše di tiilego e a oketšega ka lebaka la kgolo ya palo ya batho, toropofatšo ya lebelo le mediro ya ikonomi. Ka lebaka leo, peakanyo le phethagatšo ye e sa lekanego, ye e sego gona le ye e fokolago ya taolo ya ditlakala go la Ga-Rankuwa e feletša ka ditlakala tše ntši ka malahlelloditlakaleng. Le ge malahleloditlakala a rwala ditlakala tše dintši kudu, ditlakala tšeo di sa laolegego di ka dira gore go be le koketšego ya dikgase tša *greenhouse*, phokotšego ya bobotse bja tikologo le tšhilafatšo ya meetse le mabu a tikologo ke meetse a go hlotlwa ka ditlakala (leachate) ao a sa hlwekišwago, kudu ge ditshepedišo tše di swanetšego le tlhokomelo di sa latelwe. Ntle le gore go dirwa dinyakišišo le temošo ye kaakang ka mokgwa wo wa taolo ya ditlakala, tšhilafatšo ya tikologo e sa bonagala ka lebaka la tlhokomelo ye e sa lekanego/swanelago. Maikemišetšomagolo a nyakišišo ye e be e le go sekaseka ditlamorago tša tikologo tša bolahleloditlakala godimo ga mmu le go boleng bja meetse ka lekheišeneng la Ga-Rankuwa. Disampole tša mmu di ile tša tšewa ka pherimitareng ya bolahleloditlakala mo mahlakoreng ka moka a mane a magolo, morago sampole ya go latela ya 1 km le ya 2 km go tloga lehlakoreng la mathomo, bjalobjalo. Ditšhipi tše boima, go akaretšwa Khromiamo, Mekhuri le Morodi, di hlahlobilwe go sampole ye nngwe le ye nngwe ya tšhilafalo yeo e ka bago gona ka bolahleloditlakala. Disampole tša meetse di tšeerwe ka go tsenya ditšhelo thwii ka gare ga moela wa kgauswi go tšea disampole tša meetse (dikarolo tša noka ya ka godimo, ya bogareng bja noka le ya noka ya ka fase). Disampole di ile tša išwa laporatoring gore di hlahlobje, gomme go ile gwa dirišwa diteko tše dingwe tša dipalopalo le ditšhupetšo. Tshekatsheko ye nngwe e dirilwe ka go šomiša XLSTAT. Tshekatsheko ya (pH), Sodiamo (Na^+), Potasiamo (K^+), Kalsiamo (Ca^{2+}) le Foseforase (P) e sekasekilwe go bapa le ditšhipi tše boima go lekola letswai la meetse. Tshekatsheko ya mmu kgauswi le bolahleloditlakala e bontšhitše pH ye ntši le mohlala wa kabo ya tšhipi, ka mafelo a bodikela ao a oketšegago ge motho a katoga bolahleloditlakala (bodikela 1 km (4.8) < bodikela 2 km (5.5) < pherimita ya bodikela (6.0)), le lefelo la ka leboa 1 km (5.0) le lefelo la ka bohlabela 2 km (5.7) le bontšha esiti ye ntši ka lebaka la meetse a go hlotlwa ka ditlakala tša bolahleloditlakala. Palo ya tšhipi ye boima e be e oketšegile (pherimita ya borwa 238.095 mg/kg le leboa 1 km 55.660 mg/kg); le ge go le bjalo, dikelo ka moka di be di le ka fase ga magomo ye e dumeletšwego ya WHO (Zn (95.00 mg/kg), Mn (850.00 mg/kg) le Cu (45.00 mg/kg)) gomme di arotšwe bjalo ka "tše di sa šilafatšwago" ka fase ga Intekse ya tshekaseko ya tšhilafalo (Geo-accummulation) ya Müller. Noka ya kgauswi le bolahleloditlakala e bontšhitše maemo a alkaline ye nnyane (noka ya ka godimo (7.24) le noka ya ka fase (7.70)), eupša matlafatšo ya phepo, gagolo ka karolong ya bogareng bja noka (7.48), go dirile gore go be meetse a boleng bja go fokola. Noka e ile ya hlopša bjalo ka haepharitrofi ka lebaka la tshekatsheko ya TSI ya foseforase ka nokeng, moo dikelo ka moka tše di dirilwego ka khomphutha di bego di feta 100 (noka ya ka fase 111.14, noka ya ka godimo ll5.23, le noka ya bogare 125.49), ka merwalo ya phepo le sodiamo ye ntši, go e dira lefelo la tšhilafatšo ye ntši. Diphesente tša dikelo tša Sodiamo di ile tša dula ka gare ga magomo a go amogelega a nošetšo: noka ya ka godimo (20.3%Na), noka ya magareng (23.9%Na) le noka ya ka fase (3.6%Na). Disampole tša mmu tše di hweditšwego ka gare ga rediase ya 2 km di be di hlaelela, di laetša kotsi ya fase ya diphedinyana tše nnyane. Disampole tša meetse a noka di bontšhitše palo ye kgolo kudu ya E. coli le palomoka ya dikholifomo, kudu ka karolong ya ka fase, moo dipalo di fihlilego maemong a bohlokwa a go tshwenyega ka maphelo a setšhaba. Go šišinywa gore bašomi ba malahleloditlakala ba amogele ditharollo tša boentšenere tša sebjalebjale, go swana le ditshepedišo tša maemo a godimo tša kgoboketšo ya meetse a go hlotlwa ka ditlakala le ditshepedišo tša tlhwekišo, go fokotša tokollo ya ditlakala tše kotsi ka gare ga mmu le meetse a kgauswi. Go feta moo, kopanyo ya karoganyo ya mothopo wa ditlakala le mananeo a tšhomišoleswa a kakaretšo e swanetše go etiša pele phokotšo ya bolumo ya ditlakala tše kotsi le tša go se šomišege gape tšeo di tsenago ka gare ga malahleloditlakala.

Mantšu a bohlokwa: Tikologo, Ga-Rankuwa, ditšhipi tše boima, bolahleladitlakala, boleng bja mmu, taolo ya ditlakala, boleng bja meetse.

NKOMISO

Hi ku angarhela, vuendli bya thyaka ro tiya ya engeteleka hikwalaho ka nhlayo ya vanhu leyi engeteleke, migingiriko ya swa ikhonomi na vudoroba. Vutandzhaku, ku kunguhata ka le hansi na ku nga vi kona ku nga ringanelangi no humelerisa vuendli bya thyaka ro tiya bya mimbuyelo ya le Ga-Rankuwa bya thyaka ro tala etaleni. Hambilewi tala ri nga na thyaka ro tala, ri nga lawuleki ri nga engetela ku humeiwa ka moyakhubu exibakabakeni, ku hunguteka ka vusaseki bya mbangu, na nthyakiso wa mati ya le kusuhi na misava hi likheti yi nga endliwangiki nchumu, ngopfu loko maendlelo ya kahle no hlayisa swi nga landzeleriwangi. Handle ka ku va ndzavisiso wu nga va njhani na mpfhumba swi endliwile ka endlelo leri ra malawulelo ya thyaka, nthyakiso wa swa mabngo i vumbhoni hikwalaho ka ku va swi nga langutisisiwi kahle.

Xikongomelonkulu xa dyondzo leyi a ku ri ku hlela switandzhaku swa mbangu swa hundzuko wa misava emisaveni na nkoka wa mati elokixini ra Ga-Rankuwa. Tisampulo ta misava ti hlengeletiwile eka pherimitara ya ndhawu hi yoxe ka mathelo ya mune hinkwawo ya khompasi, ku landzeriwa sampulo ya 1km na 2 km kule ku suka ka xiphemu xo sungula, na swin'w. tisimbhi to tika, ku katsa Khuromiyamu, Mekhuri na Lidi, swi hleriwile ku ya hi sampulo ku va ku ri na vukona bya nthyakiso bya ntato wa ndhawu. Tiampulo ta mati ti hlengeletiwile hi ku celela tikhoyithenara kusuhi ku hlengeleta tisampulo ta mati (ehenhla ka nambu, exikarhi ka nambu na le henhla ka swiyenge swa le hansi ka nambu).

Tisampulo tiyisiwile elaborotari ti xoperiwa, na nhlayonhlayo wa swikambelo na tikhonsepe xiheri ti tirhisiwile. Vuxoperi byin'wana byi endliwile ku tirhisiwa XLSTAT. Nxopaxopo wa (pH), Sodium (Na^{+}), Potassium (K^{+}), Calcium (Ca^{2+}) na Phosphorus (P) swi hleriwile ka tisimbhi to tika to kambela lungho wa munyu ematini. Nhlelonhlelo wa le matini kusuhi na ntatiso wa ndhawu wu kombisa nkoka wa pH no landzelerisa wo hambana wa hangalaso wa nsimbhi, na matlhelo ya vupela-dyambu yo engeteleka loko u ya emahlweni ku suka ka ntatiso wa ndhawu (vupela-dyambu 1 km (4.8) $<$ vupeladyambu 2 km (5.5) $<$ vupela-dyambu bya pherimitara (6.0)),

Nambu lowu nga ekusuhi na ntalo wa ndhawu wu kombisa swiyimo swa alkaline nyana (ehenhla ka nambu (7.24) na le hansi ka nambu (7.70)), kambe swi fuwule hi swakudya, ngopfu eka xiyenge xa nambu wa le xikarhi (7.48), wu kongomeke ka mati ya xiyimo xa le hansi. Namu wu avanyisiwile tanihi hayiperewutirofiki hikwalaho ka nxopaxopa wa TSI ka phosphorus enambyeni, laha mikoka hinkwayo yi khompyutarayiziwile ku tlula 100 (nambu wa le hansi

111.14, nambu wa le henhla ll5.23, na nambu wa le xikarhi 125.49), wu ri na swakudya lewi engeteriweke na mindzhwalo ya sodium, swi endlaka ndhawu yikulu ya nthyakiso

Phesente ya minkoka ya Sodium yi va ka swipimelo swo amukeleka swa ncheleto: nambu wa le henhla (20.3%Na), nambu wa le xikarhi (23.9%Na) na nambu wa le hansi (3.6%Na). tisampulu ti kumekeke ka 2 wa km rhediyasi a ti kayivelekile, lewi kombisaka nxungeto wa le hansi wa swa mikhurobiyali. Tisampulu ta nambu ti kombisiweke hi nkoka ti engeterile mitsengo ya E. ya kholi na mitsengo hinkwayo ya kholifomo, ngopfu eka segimente ya le hansi, laha ku hlayela ku fikaka tilevele ta vurhon'wana to rihanyu ra vanhu. Swa bumabumaleka leswaku switirhisiwa swa ntalo wa ndhawu swi tirhisa switshunxu swo yisa emahlweni swa vuinjhiniyara, swo fana na xiyimo-vutshila xa lekheti nhlengeletanelo na tisisiteme to khoma kahle, ku hunguta humeso wa swithyakiso swa chevu eka misava na miri ya mati.

Ku yisa emahlweni, nhlangano wa nthyakiso hambano ka xihlovo na iphurogireme to twisiseka to pfuxelela ti rhangisiwa emahlweni ku hunguta ntalo wa nkala-bayodigiretebulu na mitalo yandhawu yo kombia nthaka leri nghenaka.

Marito ya nkoka: Mbangu, Ga-Rankuwa, tinsimbhi to tiya, hundzuluko wa misava, nkoka wa misava, thyaka, nkoka wa mati.

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LIST OF ACRONYMS

AGI	Acute Gastrointestinal Illness
Ca^{2+}	Calcium
Cd	Cadmium
CDCP	Centres for Disease Control and Prevention
CF	Contamination Factor
c.f.u. / cfu	Colony Forming Units
CI	Composite Index
Cu	Copper
Cr	Chromium
D	Dioxygen
DWAF	Department of Water Affairs and Forestry
ECDPC	European Centre for Disease Prevention and Control
EFTA	European Free Trade Association
EIA	Environmental Impact Assessment
EPA	Environmental Protection Agency
EPR	Extended Producer Responsibility
EU	European Union
EWQI	Entropy Weighted Water Quality Index
GHG	Greenhouse Gas
HACCP	Hazard Analysis and Critical Control Points
HPI	Heavy-metal Pollution Index
IWA	International Water Association
Igeo	Geoaccumulation Index
K^+	Potassium
mCd	Modified degree of Contamination
MLR	Multiple Linear Regression
MMC	Member of the Mayoral Committee
Mn	Manganese
MSW	Municipal Solid Waste

N	Nitrogen
Na^+	Sodium
ND	Not Detected
NEMWA	National Environmental Management: Waste Act
NG	No Growth
Ni	Nickel
NPI	Nutrient Pollution Index
NWA	National Water Act
P	Phosphorus
Pb	Lead
Pi	Percent Sodium
RCRA	Resource Conservation and Recovery Act
RSA	Republic of South Africa
S	Sulphur
SD	Secchi Disc transparency
SDG	Sustainable Development Goal
SOER	State of the Environment Report
SOHO	South African History Online
SWM	Solid Waste Management
TNTC	Too Numerous To Count
TP	Total Phosphorus
TSI	Trophic State Index
UN	United Nations
USEPA	United States Environmental Protection Agency
WHO	World Health Organisation
WM	Waste Management
WQI	Water Quality Index
Zn	Zinc

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CHAPTER 1

BACKGROUND AND INTRODUCTION

1.1 Background of the Study

The rapid growth of the global population, together with accelerating urbanisation and industrialisation, has resulted in a significant volume of solid waste generated worldwide. Earlier studies established that there is a clear relationship between population growth and waste generation, highlighting that development processes produce waste streams that require effective management (Nwokedi *et al*., 2011; Narayana, 2009). As societies continue to modernise, this trend has intensified, contributing to increased waste volumes as well as to broader environmental challenges, including air pollution, land degradation, and water contamination (Kaza *et al.,* 2018). According to recent evidence, global waste generation is expected to rise substantially in the coming decades, especially in developing countries where rapid urban expansion is hardly matched by adequate waste management infrastructure (World Bank, 2018; United Nations Environment Programme, 2021).

Sanitary landfilling is still one of the most widely used methods of solid waste disposal globally because of its relative affordability and practicality, especially in Low- and Middle-Income Countries (LMIC) (Narayana, 2009). However, although landfills serve a crucial role in waste containment, they are also associated with environmental risks when not properly designed or managed. A landfill refers to as a designated site for the controlled disposal of waste in a way intended to minimise harm to public health as well as the environment (Park *et al*., 2001; Laconi *et al.,* 2006). Regardless of these intentions, growing empirical evidence shows that landfill systems frequently fail to fully contain pollutants, especially in contexts where monitoring and regulatory enforcement are weak (Ferronato & Torretta, 2019; Njoku *et al.,* 2021).

One of the most critical environmental concerns associated with landfills is that they generate leachate. Leachate is formed when precipitation infiltrates waste material, and the various contaminants are dissolved and transported through the waste mass and into surrounding soils and water systems (Adamu *et al.,* 2017). Early studies identified a range of hazardous substances in landfill leachate, such as heavy metals, polycyclic aromatic hydrocarbons, and emerging contaminants (for example, pharmaceutical residues) (Swati *et al.,* 2014). According to more recent research, landfill leachate remains a complex and highly variable mixture of organic and inorganic pollutants, and most of the pollutants pose long-term risks to ecosystems and human health (Eggen *et al*., 2019; Naveen *et al.,* 2021).

Soil is normally identified as the most immediately affected environmental component in landfill areas, because it acts as the primary recipient of infiltrating contaminants. Earlier work suggested that percolation processes allow for the transfer of chemical substances from waste into surrounding soils (Magaji, 2012). This is supported by more recent studies, which demonstrate that landfill leachate can significantly alter soil physicochemical properties, while reducing soil fertility and introducing toxic elements that may enter the food chain (Olusola *et al*., 2019; Aderemi *et al.,* 2020; Fadhullah *et al.,* 2022). These impacts are particularly concerning in peri-urban and rural communities because land is often used for agriculture.

In addition to soil contamination, landfill leachate poses a serious threat to surface water as well as groundwater systems. Earlier studies highlighted the potential for leachate migration to contaminate water resources, posing risks to human health (Samuel & Albert, 2014; Nta & Odiong, 2017). Contemporary research further emphasises that poorly managed landfill sites are among the leading sources of groundwater pollution globally, especially in regions with no engineered liners and leachate collection systems (Mukherjee *et al*., 2020; Naveen *et al*., 2021). The persistence of contaminants in water systems increases the likelihood of bioaccumulation as well as long-term exposure, raising serious public health concerns.

In the South African context, waste management is mainly the responsibility of local government, as mandated by the Constitution of the Republic of South Africa (1996) and further supported by the National Environmental Management: Waste Act 59 of 2008. Municipalities are tasked with providing waste collection services, as well as managing disposal facilities, promoting recycling, and ensuring compliance with environmental standards. Earlier local studies pointed to challenges that include inadequate infrastructure, limited technical capacity, and weak enforcement mechanisms (Vennekens & Govender, 2005; Mamosa, 2010). According to more recent reports, these challenges persist, with many landfill sites in South Africa failing to meet minimum compliance standards, increasing the risk of environmental contamination (Department of Forestry, Fisheries and the Environment [DFFE], 2020; Godfrey & Oelofse, 2021).

Furthermore, the implementation of advanced technologies for leachate treatment and pollution control is still limited due to financial constraints coupled with capacity gaps within municipalities. Earlier observations highlighted the high costs associated with effective landfill management, and also the lack of technological adoption (Al-Muzaini *et al.,* 1995; Milad, 2014). Recent studies confirm that these barriers still hinder improvements in landfill management systems, especially in rapidly growing urban areas (Ferronato & Torretta, 2019; Godfrey & Oelofse, 2021). This is why landfill sites, if not adequately monitored and managed, may pose ongoing environmental and health risks.

Due to these challenges, there is an increasing consensus in the literature that continuous monitoring, as well as improved waste management practices and the adoption of sustainable technologies are needed for the mitigate the environmental impacts of landfill leachate. Earlier and recent studies both emphasise the importance of integrated approaches that combine regulatory enforcement, technological innovation, and community awareness for the reduction of the risks associated with landfill contamination (Babagana & Babagana, 2015; United Nations Environment Programme, 2021; Fadhullah et al., 2022).

1.1.1 Role of municipalities in waste management

To keep the environment healthy and viable, waste management is essential (Smith, 2019). Municipalities are mostly responsible for waste collection, transportation, and recycling (Johnson. 2020). The municipalities' role is to ensure waste is disposed of correctly, set up collection systems, and institute regulations for managing waste (Ludwig *et al*., 2018). As part of this, it is necessary to arrange for frequent collection, provide appropriate containers, and coordinate collection routes (Johnson, 2020). Added to the above, municipalities are responsible for waste disposal collection, which includes overseeing landfills and treatment facilities. Moreover, municipalities are instrumental in the education and promotion of waste management awareness among the general public.

1.1.2 Challenges associated with waste management

There are a number of challenges associated with waste management that have major repercussions on society, the economy, and the environment (Smith, 2018b). Waste production has increased due to rising populations, urbanisation and addition of more metropolitan areas (Jones & Lee, 2019). Urbanisation is associated with rising resource demands, which in turn lead to higher consumption rates and more waste (Jones & Lee, 2019). Improper waste management and disposal may result in pollution and health hazards, which is another challenge (Smith, 2018a). Added to the above, electronic and hazardous waste need specific disposal and treatment methods to minimise environmental contamination (Jones & Lee, 2019). Contamination of soil and water due to inadequate management of hazardous waste endangers ecosystems and human health. Smith (2018c) notes that waste management often faces obstacles related to infrastructure and financial resources.

Furthermore, significant expenditures are necessary to develop efficient systems for waste collection, transportation, and treatment. The problematic issue is that waste management infrastructure often goes unfunded or underfunded by both local governments and national governments (Smith, 2018c). Also, a lack of sustainable waste management technology promotion and public knowledge and involvement development also provide additional challenges (Jones & Lee, 2019). Education and behavioural changes are key to promoting waste reduction, recycling, and decomposition (Jones & Lee, 2019). The only way to overcome these obstacles is with a holistic strategy that incorporates public involvement, investments in education and infrastructure (Smith, 2018b; Jones & Lee, 2019).

1.1.3 Strategies used in the management of waste

Efforts to reduce waste may be made via process optimisation, product design that prioritises minimising waste, and the advocacy for materials that are either reused or recyclable (Smith, 2018c). Recycling is the practice of reusing and repurposing resources from waste in order to make new products. As a result, less energy is needed, and less greenhouse gas emissions are produced, and landfill space is reduced (Jones, 2020). According to Johnson and Williams (2019), another process is composting, which is the process of turning organic waste into compost that is high in nutrients. More technologies, such as anaerobic digestion and incineration, may be used to provide sustainable energy while also reducing waste (Brown & Harper, 2021).

The goal of waste-to-energy technology is to transform non-recyclable waste into usable energy in the form of heat or electricity. Miller *et al*. (2017) assert that in landfills, impermeable separators are employed to prevent groundwater contamination, gas collection systems are implemented to capture methane emissions, and proper waste compaction is implemented to ensure that landfills are managed effectively. One method of regulating products is the concept of Extended Producer Responsibility (EPR), which holds manufacturers to account from the moment a product is created until its final disposal. This method motivates manufacturers to develop products that are more easily recyclable or disposed of in a secure manner (Smith & Johnson, 2022). When combined and implemented effectively, the above strategies contribute to the sustainable management of waste, with the primary objectives being to mitigate environmental damage, preserve finite resources, and promote the development of a circular economy.

1.2 Problem Statement

Modern landfill systems are designed to minimise environmental harm through engineered containment measures, including liners, compaction, and leachate collection systems. Earlier research suggests that, when properly managed, landfills effectively isolate waste from surrounding ecosystems (Joshi *et al.,* 2017). However, according to more recent studies, in many developing-country contexts, landfill infrastructure normally does not meet required engineering and operational standards, and the outcome is unintended environmental consequences (Siddiqua *et al*., 2022; Njoku *et al.,* 2021).

In the literature, it is well-established that poorly managed landfills can generate leachate capable of contaminating soil and groundwater, while releasing landfill gases that affect air quality and human health (Eggen *et al.,* 2019; Mukherjee *et al.,* 2020; Naveen *et al.,* 2021). Additionally, studies have shown that communities located near landfill sites usually experience environmental and socio-economic impacts, for example, odour nuisance, reduced quality of life, and potential exposure to hazardous substances (Ferronato & Torretta, 2019; Fadhullah *et al.,* 2022). Although these impacts are widely documented at a global level, their extent, variability, and context-specific dynamics is still unevenly understood, especially in rapidly urbanising areas with constrained municipal capacity.

In the South African context, although legislative frameworks, including the National Environmental Management: Waste Act 59 of 2008 provide clear guidelines for landfill management, evidence shows that there are persistent gaps in compliance, monitoring, and enforcement (DFFE, 2020; Godfrey & Oelofse, 2021). This is why there is growing concern that some landfill sites may pose ongoing risks to surrounding communities, especially where ageing infrastructure, capacity constraints, and rapid population growth intersect.

The situation in Ga-Rankuwa reflects the above broader challenges but is insufficiently examined in empirical research. Existing empirical research highlights community complaints related to odours, smoke emissions, and declining environmental quality, suggesting there has been exposure to landfill-related pollutants (Ngobeni, 2020). However, these reports are largely anecdotal, while also lacking systematic scientific investigation into the actual extent of soil and water contamination in the area. Therefore, there is limited evidence to determine whether observed environmental and health concerns are directly linked to landfill leachate and associated processes.

This gap between reported community experiences as well as scientifically validated environmental data represents a critical limitation in current knowledge. Without such evidence, policymakers, municipalities, and environmental regulators will find it challenging to design targeted and effective mitigation strategies or to assess compliance with environmental standards.

Therefore, this study sought to address this gap by investigating the extent to which the landfill site in Ga-Rankuwa has influenced the surrounding soil and water quality. In doing so, the study contributes context-specific evidence that can inform sustainable waste management practices, strengthen environmental governance, and support the protection of vulnerable communities.

1.3 Significance of the Study

The pollution of soil, air, and water is a significant problem associated with inadequate landfill monitoring. Improper maintenance or closure of waste dumps nearing capacity poses a risk to public health. According to Hoornweg and Bhada (2012), the existence of uncollected waste creates an environment that is favourable for the growth of harmful bacteria, which in turn causes the spread of illnesses. For example, Alam and Ahmade (2013) identified around 22 ailments that were attributed to inadequate landfill management and domestic solid waste management, as recognised by the United States Public Health Services. The improper management of landfills is a matter of ecological significance, since the elements present contribute to the enhancement of greenhouse gases (Hoornweg & Bhada, 2012; Alam & Ahmade, 2013; Mohale, 2021). The South African Environmental Act 24 of 1998 states that all individuals within the country have the right to a secure, non-harmful environment for their health and well-being (RSA, 1998). However, in most rural communities, this right is compromised because of poor solid waste management systems (Alam & Ahmade, 2013). Given the rapid proliferation of ecological concerns resulting from inadequate solid waste management, it is crucial to conduct extensive research on solid waste management. Comprehensive comprehension and awareness of the social, legal, technological, environmental, and economic dimensions pertaining to solid waste are important in order to attain effective solid waste management (Hoornweg & Bhada, 2012; Mohale, 2021). In Ga-Rankuwa, there is a growing waste production and improper leachate management, both of which contribute to the formation of hostile physical and chemical characteristics in the soil. The present research aimed to evaluate the effects of landfills on the quality of soil and water in Ga-Rankuwa, South Africa.

1.4 Research Aim and Objectives

1.4.1 Aim of the study

The study aims to assess the environmental impacts of landfill sites on soil and water quality in Ga-Rankuwa township.

1.4.2 Research objectives

In the realisation of the above aim, the following objectives were used:

- To evaluate the presence of heavy metals in soils adjacent to the landfill site
- To assess the water quality from the river near the landfill site
- To evaluate the possible microbial markers present in the soil and water near the site.

1.5 Research Questions

The main research question is: What are the environmental impacts of landfill sites on soil and water quality in Ga-Rankuwa township?

The following sub-research questions were used to achieve the main research question:

- Which heavy metals are present in the soils adjacent to the landfill area?
- To what extent has the water quality from the nearby river been affected by the landfill?
- What are the possible microbial markers present in the soil and water?

1.6 Ethical Clearance

Ethical clearance has been acquired from the University of South Africa's College of Agriculture and Environmental Sciences Ethics Committee. Ethics approval reference: 2023/CAES_HREC/002 (Annexure A).

1.7 Scientific Contribution

The study contributes scientifically to risk assessments of concentration trends of heavy metals such as Zinc (Zn), Manganese (Mn), and Copper (Cu) in soil and river water close to active landfills, contributing to empirical data on regional geochemical baselines. The study also identifies how multivariate statistical approaches can be used jointly with water quality indices to address pollution sources in landfill-impacted water sources. Sustainable Development Goal (SDG) 6 seeks to achieve clean water and sanitation. This study provides integrated assessment of heavy metals mobility in soils and water contamination around Ga-Rankuwa landfill, under South African environmental conditions. Findings from this study will also help guide municipal engineers and environmental authorities in prioritising remediation, mitigating and monitoring measures while enforcing protection zones near landfill sites. Added to the above, the study refines source-pathway-receptor frameworks for areas that are semi-arid by incorporating the possible seasonal river fluctuations in dispersion of pollutants. Moreover, the study contributes theoretically and practically through the support of sustainable landfill management, informing risk-based remediation, and offering methodological insights for future research on waste-induced pollution in developing areas.

1.8 Limitations of the Study

Although this study provides valuable insights into the environmental impacts of landfill sites, some limitations are acknowledged. First, the study relied on a single sampling period, which may not capture seasonal variations in heavy metal concentrations and microbial activity. Longitudinal studies would provide a more comprehensive understanding of the temporal dynamics of contamination. Second, the study focused on a limited number of heavy metals (Zn, Mn, and Cu) and microbiological markers. A comprehensive evaluation of environmental quality would benefit from broadening the range of pollutants considered, incorporating a wider variety of contaminants, including organic pollutants and additional heavy metals. Third and last, the research was conducted in a particular geographic area (the township of Ga-Rankuwa), which could restrict the applicability of the findings to other regions that have varying environmental and socioeconomic conditions.

1.9 Chapter Outline

Chapter 1: Introduction and background

This chapter provides an introduction & background of the study, as well as the problem statement. It also describes the aim and objectives, research questions that shaped the study and ethical and scientific contributions of this study.

Chapter 2: Literature review

Chapter Two provides a comprehensive examination of global and local perspectives on waste management, emphasising the persistent environmental challenges posed by landfilling. The chapter further explores the geochemical and biological processes governing leachate formation, heavy metal mobility, and microbial activity in contaminated environments. The chapter delineates various landfill types and evaluates their environmental implications. Also analysing South Africa’s legislative landscape, including the Constitution, the National Water Act, and the Waste Act, which collectively aim to promote sustainable waste governance.

Chapter 3: Research methodology

This chapter provides a clear description of the procedures that were followed when conducting data collection and analysis. It provides a broad explanation of the study area, research design, research approach, data collection and sampling, sampling method, and analysis of the data.

Chapter 4: Research findings and interpretation of results

This chapter presents the results, which are discussed with reference to published empirical and literature studies.

Chapter Five: Conclusions and recommendations

In this chapter, the researcher summarises key findings, concludes the findings of the study and drafts recommendations that would assist the relevant stakeholders and future researchers in this field.

1.10 Chapter Summary

This chapter provided the study's background, the problem statement, and a discussion of why the study is significant. Research objectives and questions were outlined, and the scientific contribution and the limitations of the study, ending with the chapter outline for the research report. In the literature is reviewed.

CHAPTER 2

LITERATURE REVIEW

2.1 Introduction

As the world moves towards its urbanised future, the problem of solid waste continues to increase, and it becomes an issue for the environment and humanity. The focus of this literature review is to provide clarity on the findings of other researchers' disclosure within the areas impacted by the landfill, with a specific interest in the soil and water quality, as well as the strategies and action plans in combating this crisis. Also, various sections under waste management are discussed in detail.

2.2 Waste Management as a Global Problem

Waste management is no longer a predominantly local service issue but a complex global environmental challenge shaped by factors that include rapid urbanisation, industrialisation, and changing consumption patterns. Earlier studies highlighted that population growth and economic development are strongly correlated with increased waste generation, specifically in urban centres (Singh *et al.,* 2014). However, recent global assessments demonstrate that the scale and complexity of waste streams have intensified, and municipal solid waste is expected to increase significantly in low- and middle-income countries due to urban expansion as well as inadequate infrastructure (Kaza *et al.,* 2018; United Nations Environment Programme, 2021). This shift has repositioned waste management as a critical component of global environmental governance, and not purely a municipal function.

There is a dominating area of consensus in the literature that waste management systems differ significantly between developed and developing contexts. Earlier work suggested that high-income countries have transitioned toward integrated waste management systems prioritising recycling, waste minimisation, and energy recovery (Chang & Pires, 2015). However, developing countries still rely heavily on waste collection and disposal, limitedly emphasing on waste reduction or circular economy approaches (Faheem & Khan, 2015). More recent studies reinforce this divide; they argue that structural constraints (including financial limitations, governance inefficiencies, and technological gaps) are a barrier to the adoption of sustainable waste management practices in many African and Asian contexts (Ferronato & Torretta, 2019; Godfrey & Oelofse, 2021).

Regardless of technological and operational advancements, a limitation identified across the literature is the continued reliance on "end-of-pipe" solutions, particularly landfilling. Although earlier scholars viewed landfills as a necessary and cost-effective waste disposal method (Singh et al., 2014), recent research critiques this approach because it addresses symptoms and not root causes of waste generation (Kaza *et al.,* 2018). This has resulted in growing calls for a transition toward preventative and circular waste management systems prioritising waste reduction, reuse, and material recovery (United Nations Environment Programme, 2021).

From an environmental perspective, there is broad agreement that waste management significantly contributes to global ecological challenges, for example, climate change. Earlier estimates suggested that waste-related activities contribute about 5% of global greenhouse gas emissions (Mohareb *et al.,* 2011). Recent analyses confirm that landfills are still a major source of methane emissions, a potent greenhouse gas, particularly in regions where waste decomposition is unmanaged (Gautam & Agrawal, 2020; Kaza et al., 2018). However, scholars differ when it comes to the extent to which improved landfill engineering alone can mitigate these impacts. Although some argue that sanitary landfills significantly reduce environmental risks when properly managed, others contend that even engineered systems cannot fully eliminate long-term contamination risks, especially in resource-constrained settings (Ferronato & Torretta, 2019; Njoku et al., 2021).

A further area of growing scholarly attention is on the environmental risks associated with landfill leachate and contaminant transport mechanisms. Although earlier studies mainly described the presence of pollutants in landfill systems, more recent research provides a deeper understanding of how contaminants migrate through soil and groundwater systems. For example, some studies demonstrate that heavy metal mobility is influenced by soil pH, permeability, and organic matter content, which affect the rate as well as extent of contaminant transport (Eggen *et al.,* 2019; Mukherjee et al., 2020). Similarly, the formation and composition of leachate are now understood to vary depending on waste composition, as well as climatic conditions and landfill age, introducing significant variability in environmental impacts (Naveen *et al.,* 2021). These findings highlight that predicting contamination pathways is complex and highlights that there is a need for site-specific investigations.

In the African and South African context, recent studies reveal that a lot of landfill sites operate under conditions that increase the risk of environmental contamination. Although policy frameworks exist to regulate waste management, there are still persistent implementation challenges, such as inadequate monitoring, ageing infrastructure, and limited technical capacity (Adeleke *et al*., 2021; Godfrey &

Oelofse, 2021; DFFE, 2020). Empirical studies conducted in various African settings indicate elevated levels of heavy metals as well as other contaminants in soils and groundwater near landfill sites, suggesting that environmental risks are not just theoretical but actively manifesting in affected communities (Aderemi *et al.,* 2020; Njoku *et al.*, 2021).

Regardless of these advances, various critical gaps remain in the literature. First, most of the existing research is either global or regionally generalised; there are limited fine-scale, site-specific studies that examine contamination dynamics in local contexts. Second, although contaminant presence is widely documented, there is limited focus on linking environmental measurements to community-level impacts and lived experiences. Third, inconsistencies exist in methodologies used to assess soil and water contamination, which makes it difficult to compare findings across studies. These gaps highlight that there is a need for integrated, context-specific research combining environmental analysis with localised understanding of risk exposure.

The current study responds directly to these gaps because it focuses on the Ga-Rankuwa landfill site in South Africa. Examining soil and water quality in relation to landfill activities means the study can contribute empirical" evidence that bridges the divide between global knowledge and local realities. In this way, the study aligns with emerging scholarly calls for context-sensitive research that can inform policy and practice in waste management.

2.3 Conceptual Framework

This study is grounded in the Source–Pathway–Receptor (SPR) model, which is a widely used environmental risk framework conceptualising contamination as a dynamic process that links pollutant generation, transport mechanisms, and exposure outcomes. Within the context of landfill systems, the source refers to waste deposits and landfill characteristics (for example, waste composition, age, and management practices), the pathway represents the movement of contaminants through environmental media (soil and water), and the receptor includes human populations, ecosystems, and water resources exposed to contamination (Abdel-Shafy & Mansour, 2018; Njoku *et al*., 2021; Naveen *et al.,* 2021).

Unlike purely descriptive approaches, the SPR model provides a causal and analytical structure that allows this study to systematically link landfill processes to measurable environmental outcomes. Specifically, landfill-derived contaminants are mobilised through leachate formation, driven by rainfall infiltration and waste decomposition processes. These contaminants migrate through soil profiles and groundwater systems, altering physicochemical properties and potentially exceeding environmental safety thresholds (Eggen *et al.,* 2019; Mukherjee *et al.,* 2020).

This conceptual framework is operationalised through the following variables:

- Independent Variables (Source-related factors): Landfill characteristics (age, size, and management practices), Waste composition (organic vs inorganic waste), and Rainfall intensity and infiltration rates.
- Intervening Variables (Pathway processes): Leachate generation and composition, Soil permeability and structure, and Hydrogeological conditions (groundwater flow and depth).
- Dependent Variables (Environmental outcomes).
- Soil quality parameters: pH, Electrical Conductivity (EC), heavy metal concentrations (for example, Pb, Cd, and Zn).
- Water quality parameters: pH, turbidity, Total Dissolved Solids (TDS), Chemical Oxygen Demand (COD), and heavy metals.

These variables reflect the transformation of contamination from source to pathway to measurable environmental impact, allowing for the empirical testing of the framework. The conceptual framework is presented in Figure 2.1

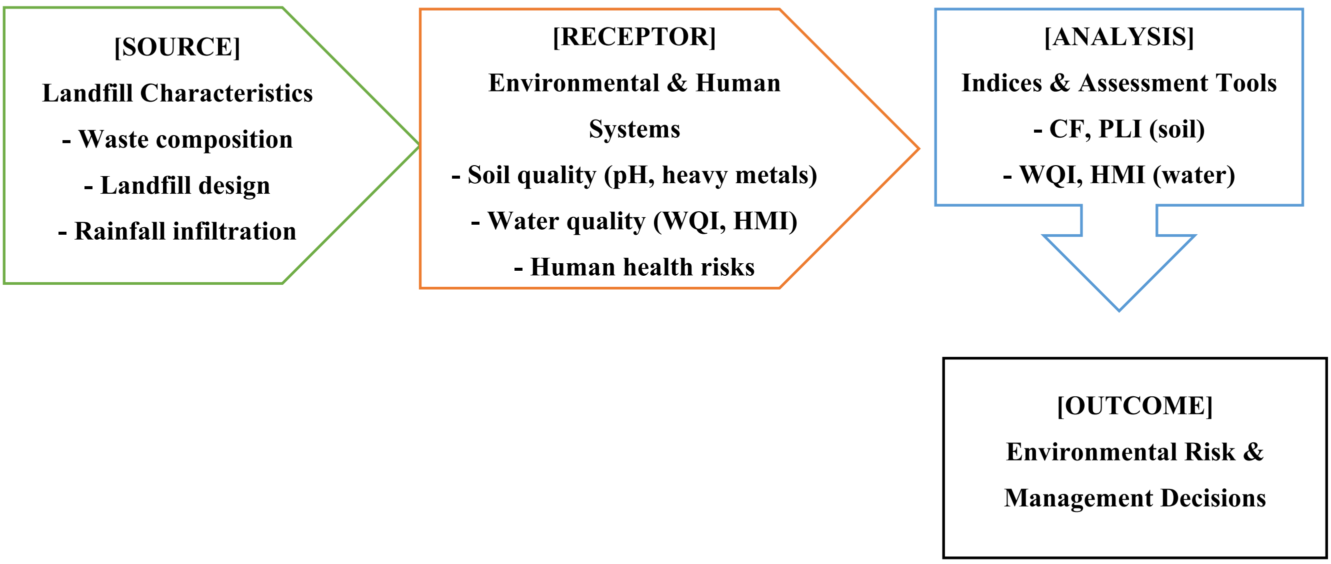

Figure 2.1: Conceptual Framework

To strengthen the analytical dimension of the study, contamination is measured directly and also interpreted using established indices:

- Contamination Factor (CF) – assesses the level of heavy metal contamination relative to background values
- Pollution Load Index (PLI) – evaluates overall soil pollution status
- Heavy Metal Index (HMI) – determines the degree of water contamination
- Water Quality Index (WQI) – integrates multiple water parameters into a single indicator of suitability

These indices provide a standardised and comparative basis for the evaluation of environmental quality and linking observed contamination levels to potential ecological and human health risks (Aderemi *et al.,* 2020; Fadhullah *et al.,* 2022).

The conceptual framework aligns with global environmental priorities, particularly those reflected in the United Nations Sustainable Development Goals (SDGs), which emphasise sustainable waste management and protection of water resources (UN, 2018). However, although such global frameworks provide normative guidance, empirical studies highlight that there are persistent implementation challenges in developing countries.

Recent African and South African research demonstrates that ineffective waste management systems, for example, poorly regulated landfills and illegal dumping, still degrade environmental quality (Godfrey & Oelofse, 2021; Naidoo *et al.*, 2024). For example, studies in South African municipalities reveal that landfill-related leachate contributes to soil and groundwater contamination, especially in areas with limited monitoring and infrastructure (Viljoen *et al.*, 2021). These findings reinforce the relevance of the SPR model when it comes to understanding how systemic weaknesses translate into measurable environmental risks.

2.4 Landfilling as a Waste Management Strategy

Landfilling is one of the most widely used waste management strategies globally, especially in developing countries where there are limited alternative waste treatment technologies. Earlier studies framed landfills as a cost-effective and practical solution for the management of increasing volumes of municipal solid waste (Thomsen *et al.,* 2012). However, more recent research challenges this position because it demonstrates that the environmental impacts of landfills are far more complex and context-dependent than previously assumed.

A growing body of contemporary literature adopts Life Cycle Assessment (LCA) approaches to evaluate landfill impacts more comprehensively. For example, Sauvé and Van Acker's (2020) landfill life cycle assessment in Europe shows that even well-engineered landfill systems in Europe contribute significantly to long-term environmental burdens like greenhouse gas emissions, leachate production, and resource inefficiencies. The study's findings suggest that landfills should no longer be viewed as neutral disposal options but as active environmental systems with long-term ecological footprints. This marks a shift from earlier perspectives that emphasised engineering containment as sufficient mitigation.

Recent empirical studies provide strong evidence of landfill-related groundwater contamination. Przydatek and Kanownik's (2019) groundwater landfill study found that even small, localised landfill sites can significantly alter groundwater quality, with elevated concentrations of nitrates, heavy metals, and organic pollutants. This finding challenges the assumption that environmental risks are limited to large-scale or poorly managed landfill facilities. On the other hand, comparative research from North Africa demonstrates variability in contamination levels depending on landfill design and management practices. For example, Benaddi *et al.* (2022) show that differences in liner systems, waste composition, and hydrological conditions result in markedly different groundwater contamination outcomes across landfill sites.

Earlier large-scale reviews, for example, Han *et al.* (2016), identified landfill leachate as a primary driver of groundwater pollution, especially in rapidly urbanising regions. Recent studies build on this by explaining mechanisms of contaminant transport, and they show that pollutant mobility is influenced by various factors, including soil permeability, climatic conditions, and landfill age (Eggen *et al.,* 2019; Mukherjee *et al.,* 2020). These findings highlight that contamination is not a uniform process; it is a site-specific interaction between environmental and anthropogenic factors, and this reinforces the need for localised studies such as the present one.

At a policy level, trends in landfill use are also reflective of shifting global priorities. Data from the European Environment Agency indicates that landfill reliance in the EU-27 declined from an estimated 23% in 2010 to about 17% in 2022, which reflects a transition toward recycling and circular economy strategies. Although this suggests progress, it also raises important questions concerning data reliability and comparability. Waste reporting systems differ across countries, and indicators such as "landfill rate" may mask informal dumping and variations in classification methods. This is why scholars caution that global landfill statistics should be interpreted critically, especially when applied to developing-country contexts where data systems are less robust (Kaza *et al.,* 2018; Ferronato & Torretta, 2019).

Regardless of the declining landfill rates in some high-income regions, landfills are still dominant in many African and developing contexts, where infrastructure limitations constrain the adoption of alternative waste management strategies. In these settings, landfills are usually poorly regulated or inadequately engineered, which increases the likelihood of leachate migration and environmental contamination (Godfrey & Oelofse, 2021; Njoku *et al.,* 2021). This creates a contradiction in the literature: although landfills are decreasing in importance in some regions, they remain central (and environmentally risky) in others.

A critical gap emerging from this body of research is that there is a limited number of integrated, site-specific studies that simultaneously assess soil and groundwater contamination in relation to landfill activity, especially in South African contexts. Although international studies provide valuable insights into contamination mechanisms and risks, their findings cannot be directly generalised because of differences in climate, geology, waste composition, and regulatory enforcement.

Therefore, this study builds on existing literature because it provides context-specific empirical evidence from the Ga-Rankuwa landfill site. Analysing both soil and water quality using established indices means that the study contributes to a deeper understanding of landfill impacts in a South African setting, addressing the gap between global knowledge and local environmental realities.

2.5 Pollution of Soil and Water from Improper Landfill Management

Globally, a substantial body of research demonstrates that landfill leachate is the main driver of soil and groundwater contamination, particularly where containment systems are inadequate. According to earlier reviews, leachate formation is governed by rainfall infiltration, waste composition, and landfill age, resulting in complex mixtures of organic pollutants, nutrients, and heavy metals (Han *et al.,* 2016). More recent studies provide deeper insight into contaminant transport mechanisms, they show that heavy metal mobility is strongly influenced by soil pH, as well as redox conditions and organic matter content, which regulate solubility and migration into groundwater systems (Eggen *et al.,* 2019; Mukherjee *et al*., 2020). Empirical studies across Europe and Asia further confirm that even engineered landfills can contribute to long-term environmental burdens. For example, Sauvé and Van Acker (2020) demonstrate through life cycle assessment that landfill systems still generate persistent emissions and leachate-related risks despite technological improvements. Similarly, Przydatek and Kanownik's (2019) groundwater landfill study found significant deterioration in groundwater quality near municipal landfill sites, with elevated concentrations of nitrates and heavy metals. There is comparative evidence from North Africa that shows variability in contamination depending on landfill design and hydrogeological conditions, with poorly managed sites exhibiting higher pollutant loads (Benaddi *et al*., 2022). Collectively, these studies highlight a consistent global pattern: landfill pollution is widespread and also highly context-dependent, shaped by environmental conditions as well as management practices.

Regionally and nationally, similar trends are evident, although sometimes exacerbated by governance and infrastructure constraints. African studies report elevated levels of heavy metals and physicochemical alterations in soils and groundwater surrounding landfill sites, particularly in areas lacking engineered liners and monitoring systems (Njoku *et al*., 2021; Aderemi *et al.,* 2020). In the South African context, research indicates that many landfill sites operate below required environmental standards, which increases the risk of leachate migration into surrounding ecosystems (Godfrey & Oelofse, 2021; DFFE, 2020). Localised studies further show that soil acts as a sink as well as a secondary source of contamination, where sorption processes initially retain heavy metals but may later release them under changing environmental conditions (Wdowczyk & Szymańska-Pulikowska, 2020). However, regardless of the growing evidence of contamination, there is still a limited number of integrated, site-specific studies that simultaneously assess soil and groundwater quality in South African landfill contexts, especially at the community level. This gap is especially relevant in areas where anecdotal evidence of pollution exists, but there is no systematic scientific validation.

2.6 Retention Mechanisms for Contaminated Soils

Contaminants such as heavy metals in groundwater are mostly controlled by soil retention responses (Wdowczyk & Szymańska-Pulikowska, 2020). The hazardous nature of metal particles is linked to their mobility in soil, which can have fatal effects on plants, animals, and humans. It is important to note that heavy metal mobility is dependent on the soil's qualities, and the hazard of these metals rises as mobility increases (Miland, 2014). It is through the process of sorption that metal particles, which have a positive charge, are attracted to soil and other natural matter particles with a negative charge (Miland, 2014). Metals bonded to solids are in equilibrium with metals in soil water because of this reversible interaction. This means that soil water is expelling tightly bound metal particles, making them less versatile than particles with weaker holdings (Miland, 2014). Heavy soils, or soils with a high muck content, have long been thought to tend to immobilise heavy metals. Since clayey soils may be disposed of safely if they are maintained by an environmental agency, waste disposal organisations can rest easy (Miland 2014).

Lead (Pb), Cu, and Zn retention in three South Wales soil types was studied by Yong *et al*. (2001). Column experiments using soils from four different locations in South Wales were used to measure metal concentration in the leachate, and Yong *et al*. (2001) found no breakthrough for Pb, Cu or Zn in any of the four soil types they tested. As the pore volume grows, the permeability stays the same or slightly rises. These soils had pH values between 7.5 and 9.5, which indicates that all heavy metals have precipitated in the soil column. Under a constant 10 kPa air pressure, the leaching tests were carried out. Leachate from a Municipal Solid Waste (MSW) dump was used for the first 2 pore volumes of the column test, followed by distilled water. Analysis of the discharge leachate showed that only a modest number of heavy metals leached into the effluent during the 5-pore volume of leaching with the test leachate were found in the three soils studied (Miland, 2014).

Additionally, Löv *et al*. (2019) used a leaching column test proposed by Zuhairi *et al*. (2008) to estimate the retention duration of heavy metals (Pb, Cu, Ni, and Zn) in Malaysian natural soils from the Selangor region. As the number of pore volumes grew, so did the relative concentration of heavy metals, with Ni and Zn being the most mobile and the sorption of heavy metals being high, as shown by the breakthrough curves. Moreover, in a column test by Löv *et al*. (2019), leachate with a very high acidity level was shown to interact effectively with the natural pH soil at the top of the column, however, this study was conducted under controlled laboratory conditions in Sweden and is not specific to Africa or South Africa, but rather demonstrates a general soil buffering mechanism.

2.7 Different Kinds of Landfills

The types of waste disposal facilities often used include sanitary landfills, MSW dumps, construction and demolition waste landfills, and industrial waste landfills. These are the four primary categories that may be used to describe a landfill. Sanitary landfills are the most common form of landfill.

2.7.1 Landfills for sanitary waste

In this type of landfills, clay is used as a liner so that waste may be contained and kept apart from the surrounding environment (Milosevic, 2012; Miland, 2014). In regions where it is essential that garbage that has been disposed be kept apart from the surrounding environment until it can be established that the region does not pose a threat to human health, sanitary landfills are utilized. After the trash has been entirely decomposed chemically, physically, and biologically, it can then be regarded safe to dispose of (Diamantis, 2013; Miland, 2014). In order to stop the seepage of hazardous chemicals into nearby soils and water sources, sanitary landfills utilize the most up-to-date leak prevention technologies. There are two primary categories of processes that are utilized in sanitary landfills (Bella *et al*. 2011). The first is known as the trench technique, while the second is known as the area method. The trench approach is favoured over the area method in regions with less garbage. Both approaches (Figure 2.2) make use of the cell concept, which involves covering garbage with dirt after it has been compacted (Thomsen *et al*., 2012; Miland, 2014).

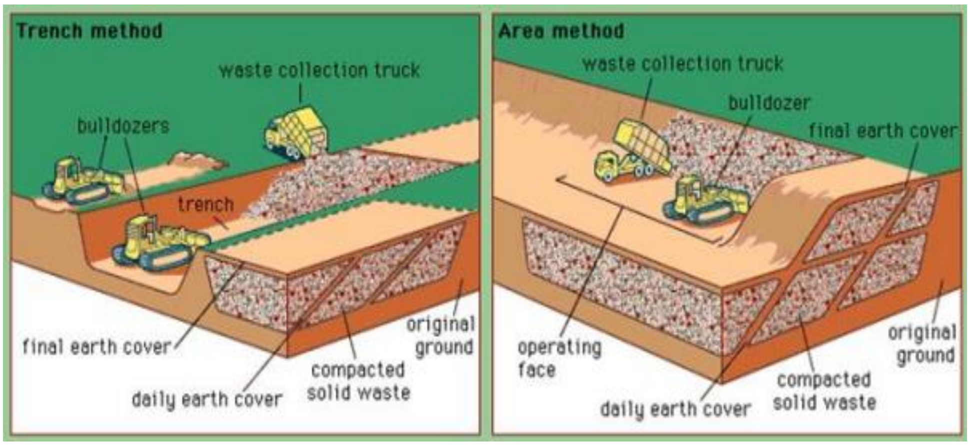

Figure 2.2: Sanitary landfill methods (Source: Nijarabi, 2010)

2.7.2 Landfills for Municipal Solid Waste

This type of landfill is equipped with a liner that is constructed with synthetic plastic so that rubbish is kept apart from the environment around it (Miland, 2014). The local and state governments are responsible for its collection and maintenance, and it is utilised as a storage facility for the garbage collected from residential areas (Geismar, 2014). The Environmental Protection Agency (EPA) has defined the substances that are permitted to be disposed of at MSW sites. Some items, including paints, chemicals, batteries, cleansers, motor oil, and pesticides, are not permitted to be thrown away at these sites (Al-Jarallah & Aleisa, 2014; Miland, 2014). A MSW site is not permitted to be used for the disposal of hazardous wastes such as bulk liquids or wastes that comprise loose liquids, garden rubbish, or scrap tyres; however, certain types of household appliances can be securely thrown there (Thomsen *et al*., 2012; Miland, 2014).

2.7.3 Landfills for waste generated during construction and demolition

The process of disposing of materials that were used in the construction, refurbishment, and demolition of buildings, roads, and bridges makes use of the processes of both construction and demolition. Most of these wastes are made up of gypsum, wood, asphalt, bricks, earth rock, glass, concrete, trees, and other components of structures. Other types of waste include earth and rock (Geismar, 2014; Miland 2014). When burned, they can release hazardous chemicals into the atmosphere and contribute to the degradation of the environment. It is imperative that the operation, siting, design, closure, and post closure standards for construction and demolition landfills be met. Even debris that have been chopped up cannot be accepted under any circumstances (Milosevic 2012). Keeping an accurate estimate of the raw materials that are required for building projects is the most effective strategy to minimise these wastes (Miland 2014). Recycling wastes of this kind not only helps save money but also helps cut down on the quantity of garbage that is dumped in landfills and helps save associated financial costs (Thomsen *et al*. 2012; Miland 2014; Wang *et al*., 2018; Shooshtarian *et al*., 2020).

2.8 Leachate as the Main Contaminant from Landfill Sites

The chemical makeup of the waste and metabolic activities inside a landfill contribute to the formation of leachate. Unwanted elements percolating into nearby soils and groundwater from a landfill or disposal site and contaminating the surrounding area are well-defined as landfill leachate (Babagana & Babagana, 2015). Solutes and solids are extracted from the waste as the liquid moves through the waste (Miland, 2014). As the leachate goes through different types of waste and biodegradation levels, the amount of contamination increases or decreases (Miland, 2014). When precipitation rate rises, so do leachate levels, especially during the rainy seasons.

2.8.1 Landfill leachate production

Landfill leachate production is a major environmental issue that arises from the disposal of solid waste in landfills. The quantity of leachate generated inside a landfill is contingent upon several parameters, such as the size and constituents of the waste, precipitation levels, and the permeability of the soil and underlying geological structures (Bjerg *et al.,* 2014). Because effective management of landfill leachate is essential to prevent pollution and protect the environment, common strategies required for leachate management include on-site treatment, off-site disposal, and recirculation (Mao *et al*., 2019). On-site treatment involves treating the leachate on-site using physical, chemical, or biological treatment methods to reduce contaminants.

As leachate migrates from waste sites, it releases contaminants that represent a substantial danger to those who obtain their drinking water directly from groundwater sources (Samuel & Albert, 2014). In describing leachate, volumetric flow rate and leachate composition are the two most essential criteria, according to Bjerg *et al*. (2014). Rainfall, surface runoff, and groundwater intrusion are all factors that affect leachate flow rate in landfills. The number of organics and ammonia in landfills differ significantly depending on how long they have been there (Peyravi *et al*., 2016). The degradation potential and contaminant concentration of landfill leachate typically decrease over time as the waste stabilizes. Initially, fresh leachate is rich in organic matter, ammonia, and other soluble pollutants, but as microbial activity and natural attenuation processes progress, these concentrations decline, resulting in a less toxic effluent in older landfill cells (Kjeldsen et al., 2002; Renou *et al.,* 2008). An old leachate contains organic chemicals with a varied array of molecular weight fractions that have complicated structures incorporating functional groups containing Nitrogen (N), Sulphur (S), and Dioxygen (O_2) (Calace *et al*., 2015; El-Salam & Ismail, 2013). As a result, the management decision may be generalised, and the treatment method stretched out based on the landfill's age (El-Salam & Ismail, 2013).

2.9 Environmental and public health significance of landfill contaminants

It is important to note that heavy metal mobility is dependent on the soil's qualities, and the hazard of these metals rises as mobility increases (Miland, 2014). Leachate from municipal solid waste landfills was shown to have a strong attraction for metals including Pb, Zn, Cu, Ni and Cd, which facilitated their migration in leachate-contaminated rivers abdel Contaminants can be transported more easily via groundwater streams if they are sorbent (assimilated/adsorbed or desorbent), which can be caused by sorption or desorption of solid particles, such as residue due to sorption. Pollutants may also be missed in water treatment systems (Lin & Chan, 2000). Testing ensures that landfill operators are operating within the constraints that have been authorised for them and that environmental regulations and standards are being followed (Smith, 2018a). It helps assess the purity of water samples, enabling early detection of potential pollution and implementing precautionary measures (Smith, 2018a). Monitoring water samples near landfills also helps assess waste management practices and infrastructure, ensuring landfill design, liners, and leachate collection systems are functioning effectively (Smith, 2018a). Sampling water in landfills provides vital information for scientific research and monitoring programs, allowing researchers to investigate pollutants' movement, assess long-term effects on ecosystems, and develop enhanced waste management practices (Smith, 2018a).

Testing water samples from landfills is paramount due to their ecological and well-being implications. Monitoring and analysing water samples from landfills is therefore essential for understanding the extent of pollution, assessing potential hazards, and implementing effective mitigation measures. Regular testing aids in the detection of hazardous contaminants, such as heavy metals, organic pollutants, and pathogens. These contaminants can infiltrate into nearby water bodies and soil, posing a threat to aquatic life, plants, and the food chain (EPA, 2020; WHO, 2017). Testing water samples ensures that landfill operations comply with environmental regulations and standards. To protect water quality, regulatory authorities frequently impose limits on the concentration of specific pollutants in leachate (EPA, 2020; WHO, 2017). Landfills still produce leachate long after they have been closed; therefore, through long-term monitoring of water samples, authorities can trace compositional changes in leachate and take corrective action if contamination levels increase. Documenting potential contamination emanating from a landfill through testing is essential for determining liability and accountability in the event of legal disputes (EPA, 2020; WHO, 2017).

Smith and Johnson (2018) prominences that regular water sample monitoring in landfills enables the early detection of contaminants such as heavy metals and organic compounds. These contaminants may migrate through the soil and contaminate adjacent water sources, endangering ecosystems and human health. Benson *et al.* (2019) state that routine water sample analysis assists in identifying the types and concentrations of pollutants present, thereby providing valuable information for regulatory compliance and remediation efforts. Water samples are typically collected from monitoring wells, surface water bodies, and other relevant locations near landfills for the purpose of assessing water quality. To quantify the presence of contaminants, analytical techniques such as chromatography, spectrometry, and microbiological assays are utilised (Smith & Weber, 2018). These analyses contribute to informed decision-making, allowing regulatory bodies and landfill operators to take the necessary measures to prevent further contamination and protect water resources.

Testing for microbial markers in landfills is crucial because it permits the monitoring and evaluation of various environmental and operational aspects of landfills. Markers of microorganisms, such as particular varieties of bacteria or microbial communities, can provide valuable information regarding decomposition processes, waste degradation, and the overall health of the landfill ecosystem. This information is essential for effective landfill management and environmental protection. To gauge the effectiveness and pace of biodegradation in landfills, biomarkers are used. Landfill operators can optimise placement and compaction processes by monitoring the diversity and composition of microbial communities, which gives researchers insight into the development of garbage breakdown.

For instance, according to Smith *et al*. (2017), gas collection tactics might be impacted by the presence of microbial groups such as methanogenic archaea, which can suggest anaerobic digestion and methane generation.

Effluent generation and groundwater pollution are two examples of possible environmental risks that may be detected with the use of microbial markers. By monitoring how microbial populations evolve, we may learn about changes in landfill conditions like pH and moisture levels, which can affect the makeup and movement of leachate. Important steps must be taken promptly to mitigate any harm to local ecosystems and water supplies, and this data is essential for that purpose. This will help generate renewable energy and decrease reliance on fossil fuels (Paritosh *et al*., 2018).

2.10 Microbial Structure in Soils and Water

The soil microbial community is influenced by various variables, such as management techniques and global change. The participation of soil microbial community in nutrient cycling makes them crucial for ecosystem functioning. The destiny of total soil microbial biomass is regulated by important processes namely: changes in biomass creation and destruction caused by physical changes, predator organisms, or microbial mortality (Bapiri *et al*., 2010). In their study, Kaur *et al*. (2005) found that environmental stress leads to an increase in bio-membrane fluidity. This causes the bilayer to break apart and results in a loss of membrane specificity in terms of permeability. Further, fatty acid composition/structure undergoes changes in response to heavy metals (HMs) toxicity, starvation, rising temperature, osmotic stress, and lowering pH (Kaur *et al*., 2005).

The presence of certain metals may influence the composition the diversity, and abundance of soil microbes. The composition of microbial communities determines their ability to withstand and recover from disturbances (De Vries *et al.* 2012). Environmental factors might also have the ability to modify the fatty acid content of the microbial membrane. Although several authors have examined and assessed the impacts of pollutants on soil microbial communities, the comprehensive understanding of the long-term consequences of the continual disposal of MSW on these populations remains incomplete (Wang *et al*., 2017; Salam & Varma, 2019; Yang *et al*., 2024). There is a notable lack of understanding about the stability of microbial communities, which refers to their capacity to withstand and recover from disruptions induced by MSW polluted with both organic and inorganic contaminants.

Waterborne pathogens and their related diseases are a major public health concern worldwide (Ramírez-Castillo *et al.,* 2015). It goes without saying that the presence of harmful microbes in water supplies is a major issue for public health; however, studies reveal that waterborne outbreaks have considerably declined over the past 20 years (Jacobsen & Koopman, 2004; Levy *et al.,* 2018; Barrett, 2019; Gharpure *et al*., 2019). Nature is full of complex microbial communities called biofilms, which cause issues in many fields, including health and ecology (Szymanska 2003). Acute gastrointestinal infections, acute respiratory illnesses, hepatitis, and multiple fatalities have been linked to waterborne microbial agents such as *Salmonella typhimurium, Vibrio cholerae, Legionella, E. coli O157:H7, and Pseudomonas* (Ramírez-Castillo *et al.,* 2015). In settings where waterborne microbial agents are present, sporadic diseases are common, and the causes of these diseases are seldom identified (Cowden 1992).

2.11 Strategies for Monitoring and Treatment of Landfill Leachate Migration

There are several variables that influence subterranean water pollution, including water table depth, soil permeability and unsaturation, effective infiltration, humidity, and the absence of suitable drainage systems (Smahi *et al*., 2013). It is common for leachate trails to contain large amounts of carbon-based compounds such as fatty acids that are not stable and have humid-like combinations (El-Salam & Ismail, 2013). The removal of leachate from the environment by re-infusion and discharge into a municipal water treatment plant is now the most extensively utilised leachate treatment procedure (El-Salam & Ismail, 2013). Leachate degradation gets increased, and landfill gas is produced because of this method's effectiveness (Bjerg *et al*., 2014). Even though methane and carbon dioxide make up the bulk of landfill gas, it nevertheless provides a potential fuel asset because of its fundamental ingredients (Lebron *et al*., 2021; El-Salam & Ismail, 2013). On-site treatment or transport of leachate to specialised treatment facilities can also be done (Bjerg *et al.,* 2014). To break down the organic compounds, activated sludge is commonly used. However, this treatment is often not suitable for the environment and is transferred to nearby sewers (Bjerg *et al*., 2014; El-Salam & Ismail, 2013). Because old and neglected landfills lack impermeable covers, they must be addressed as part of a comprehensive strategy for dealing with leachate treatment issues (Deng 2006; El-Salam & Ismail 2013). Contamination of the soil can occur because of both natural and man-made materials found on these historical sites. Organic decay could happen for up to 40 years after the facility has been shut down, causing highly polluted leachate to form when rainwater soaks into the ground (Bjerg *et al*., 2014; El-Salam & Ismail, 2013). Landfill leachate treatment often involves a combination of physical, chemical, and biological methods (Peyravi *et al*., 2016). baba. Eco-technological treatment approaches, on the other hand, are in high demand among South African landfill leachate operators, who are looking for less expensive options that are also effective enough to meet the Environmental Quality Act's operating standards (Peyravi *et al.,* 2016).

2.11.1 Four types of leachate treatments

a) Leachate transfer

The leachate transfer is based on the combination of the recycling and mixing with domestic sewage waste for treatment (Abdel-Shafy *et al.,* 2024). This is because leachate transfer alone takes long to reduce the efficiency of the treatment and increase in the effluent concentration of the biodegradation and heavy metals of the organic inhibitory compounds (Carvajal-Florez & Cardona-Gallo, 2019). Due to the high demand to control the mutual concertation of nitrogen in the leachate and high concertation of phosphorus in the sewage, the combination of recycling and mixing sewage method was suggested to speed up the leachate transfer treatment. However, this application also depends on the sequence of phases such as anoxic, filling, oxic and settling (Contrera *et al.,* 2014). Researchers demonstrated that increasing the ratio of landfill leachate to domestic wastewater resulted in COD and NH4+ -N reduction, while the addition of powdered activated carbon (PAC) could significantly improve the effluent quality (Deng *et al.,* 2018).

b) Biological treatment of leachate

This treatment is considered as one of the highly cost-effectiveness, reliable, simple and it is commonly used for the removal of leachate loaded within high levels of BOD/COD mostly at the ration of >0.5. Thus, BOD/COD of >0.5 ratio is significantly useful to eliminate nitrogenous and organic matter (OM) during immature leachate stage (Maia *et al.,* 2015). However, the effectiveness of this treatment can be reduced by the existence and the concentration of refractory compounds such as humic and fulvic acids (Abdel-Shafy *et al.,* 2024). Moreover, this treatment is vital to mitigate or minimise the biodegradation of soil microorganism under aerobic conditions through the degradation of organic compounds to form carbon dioxide emission, sludge and biogas which involves CO_2 and CH_4 (Abdel-Shafy *et al.,* 2014).

c) Chemical and physical treatment of leachate

This treatment is mostly applied as a pre-treatment or at the last purification stage and it is commonly used to remove pollutants such as stripping for ammonia (Abdel-Shafy *et al.,* 2024). The processes used in this treatment include adsorption, air stripping or coagulation/flocculation, reduction of toxic compounds, floating material, colloidal particles, suspended solids and colour through chemical oxidation (Abdel-Shafy *et al.,* 2024).

d) Membrane technologies for leachate treatment

This treatment technology has more compliance with water quality regulations in various countries according to by Abdel-Shafy *et al.* (2024). The processes involved in this treatment are membrane bioreactor (MBR), micro-filtration, ultrafiltration, Nano-filtration and reverse osmosis (Abdel-Shafy *et al.,* 2024). When the bioreactor is combined with the membrane technology, they provide a vital compact system that results in releasing ow sludge out of high biomass concentration and produce high effluent quality (Ahmed & Lan, 2012; Chen et al., 2020). According to Abdel-Shafy *et al.* (2019), the Ultrafiltration-biologically active carbon (UF-BAC) hybrid membrane bioreactor technology amalgamates biodegradation, adsorption and membrane filtration. This process had yielded about 95-98% of total organic carbon reduction (Abdel-Shafy *et al.,* 2024). The Ultrafiltration process is commonly used to eliminate the macromolecules and particles and was reported highly effective (Abdel-Shafy *et al.,* 2024) while the adsorption process depends on the adsorption by activated carbon together with biological treatment (Abdel-Shafy *et al.,* 2024). It is recorded that the greater reduction in COD levels could be accomplished by the application of adsorption treatment than the chemical methods (Silva *et al*., 2014). The chemical oxidation process can be used on old or well-stabilised leachate for the oxidation of organic matter (OM) to resulting into the most stable oxidation state of water and carbon dioxide or to the full mineralisation (Abdel-Shafy *et al.,* 2024). This process is also useful in the improvement of the biodegradation of recalcitrant organic pollutants for subsequent better economical biological treatment.

2.12 Legislation Governing Waste Management in South Africa

The Municipal Solid Waste (MSW) sector is made up of companies that provide garbage collection services for residential, commercial, and institutional properties (Viljoen *et al*., 2021). Mixed-category and composite waste is made up of a variety of different materials, including metals, glass, plastics, papers, and wood (Viljoen *et al*., 2021). What is not recycled is deposited, compacted, and covered in landfills, which are specially designed and planned sites (Miland, 2014). Hence why regulations and compliance with federal law, such as Resource Conservation and Recovery Act (RCRA) are required. These rules, or comparable government regulations, often include guidelines for placement limits, requirements for composite liners, leachate collection and removal systems, operating methods, and groundwater monitoring requirements (EPA, 2014). The EPA is guided by the Clean Air Act, and they make amendments continuously as per section 111 of the Act; a need as Green House Gas (GHG) emission levels for landfills are adjusted, for example. The EPA reviews the standards and guidelines by incorporating new data and information as they receive it (EPA, 2018). The details, however, are beyond the scope of this study because the regulations do not only differ from that of developing countries but there are notable limitations of resources in developing nations such as South Africa; hence they might not be relatable to the African context.

Van Niekerk and Weghmann (2019) reported that most waste in Africa is disposed of through dumping sites that are either controlled or not, and that only an estimated 30% is disposed of through landfills. What is alarming is that most African countries only have one landfill for each city, which is why there are a lot of illegal dumping areas and informal landfills (Van Niekerk & Weghmann, 2019). Further, legal landfills are mostly mismanaged and pose a health hazard for neighbouring communities, especially children, the disabled, the elderly and women. According to DEFF (2021:12), South Africa does not only have challenges with landfill space but also with the operation of them. Added to that, it is overwhelming for municipalities due to the financial burden involved in erecting new landfills, the resistance of communities, the reality that the land used for landfills can no longer be used productively, and the issue of GHG emissions when they cannot be effectively captured at landfills.

The development and periodic evaluation of the National Waste Management Strategy (NWMS) 2020, whose primary objective is to divert waste from landfills, are motivated by the aforementioned factors. The plan entails the development of a secondary resource economy, which is centred on waste beneficiation as part of the circular economy. In addition, composting and energy recovery are used to treat and recover soil nutrients and energy from organic waste.

2.12.1 The South African Constitution Act

The Constitution of the Republic of South Africa Act (Act 108 of 1996) has many provisions concerning the ecosystem. Important ramifications of Section 24 of the Act include the protection of people's health and welfare and the imposition of duties on the South African state, companies, and all residents. These obligations pertain to the prevention of pollution and other forms of environmental damage, as well as the promotion of conservation and sustainable development. It is incumbent upon all governmental entities, whether they at the national, provincial, or municipal level, to undertake appropriate actions, both in their current responsibilities and future endeavours, to mitigate pollution, foster conservation efforts, and uphold the principles of sustainable development.

2.12.2 Hazardous Substances Act

The Hazardous Substances Act (No. 5 1973) regulates substances that can cause harm to humans due to toxic chemicals, heavy metals, or pressure generation. The Act also regulates electronic products and categorizes them into groups based on danger level.

2.12.3 Health Act

The Health Act (No. 63 of 1977) aims to provide provisions for the enhancement of the well-being of the inhabitants of the Republic. These gases contribute to climate change and respiratory diseases. These pollutants can cause unpleasant odours, anxiety, and vertigo, thereby affecting mental health and quality of life. In addition, improper management of landfills can attract pests such as rodents, insects, and birds, which can transport and spread diseases such as bacteria, viruses, and parasites. These odours can expose adjacent communities to health dangers, including dengue fever, malaria, and West Nile virus.

2.12.4 Environment Conservation Act

The Environment Conservation Act (No. 73 of 1989) is meant, among other things, to protect the environment and make sure it is used in a controlled way. The Environment Conservation Act established a regulatory framework for landfills. Authorities are obligated to provide landfill licenses or permits under the Act, which necessitates that operators get permissions and comply with limitations. Some landfill projects must undergo an Environmental Impact Assessment (EIA) in order to identify, evaluate, and control any negative effects on the environment. The goal of the Act is to encourage ecologically responsible methods, limit negative consequences, and avoid pollution; it oversees garbage processing, transportation, and disposal. Evaluating compliance with environmental regulations and identifying possible threats requires regular monitoring and reporting. The Act also addresses landfill closure and rehabilitation by defining site restoration requirements and measures to reduce long-term environmental and health hazards.

2.12.5 National Water Act

The National Water Act, 1998 (No. 36 of 1998) (NWA) is South Africa's main law for protecting, preserving, developing, and managing water resources. In the preamble to the NWA, there is a human rights approach principle that says the goal of water resource management is to use water in a way that is good for everyone. The NWA aims to promote sustainable management of water resources, including quantity and quality. The Act requires water consumers to obtain water use authorizations to ensure responsible and sustainable water use, and noncompliance can result in revoked water use authorizations. Improper landfill management can lead to the loss of biodiversity and ecological imbalances, compromising the Act's objectives of water resource conservation, protection, and equitable allocation.

2.12.6 Air Quality Act

In order to protect the environment, the main goal of the Air Quality Act (No. 39 of 2004) was to establish new laws on air quality. Air pollution imposes significant social, economic, and environmental burdens, often left unaccounted for by those responsible for the pollution. Air pollution has considerable social, economic, and environmental costs, often borne by society rather than the polluters. It emits deleterious compounds and greenhouse gases, underscoring the fundamental right to a healthy environment. While minimising pollution through rigorous cleaner technologies, control and cleaner production practices is essential for improving air quality, Furthermore, legislative measures are necessary to enhance the government's environmental protection plans, with a special focus on improving the quality of the surrounding atmosphere, to ensure a non-hazardous environment for human health and well-being.

2.12.7 National Environmental Management: Waste Act, as amended by NEMWA

The primary objective of the National Environmental Management: Waste Act (No. 59 of 2008) is to enact legal reforms about waste management to safeguard public health and the environment. This Act is an attempt to achieve that goal by enacting concrete steps to curb pollution and environmental degradation. In addition to addressing concerns about institutional frameworks and planning factors, the Act promotes investigations into the implementation of policies to guarantee ecologically sustainable growth. Added to the above, the Act establishes national norms and standards for the regulation of waste management by governmental entities at all levels.

2.13 Chapter Summary

The literature review emphasises the global and local issues of waste management, underscoring the environmental, health, and social ramifications. Accelerated urbanisation and unsustainable consumption practices have exacerbated solid waste production, with landfilling being the predominant disposal technique. Developed nations have progressed towards integrated landfill systems; nevertheless, poorer countries such as South Africa continue to encounter challenges including mismanaged landfills, unlawful dumping, and insufficient infrastructure. Landfill leachate, including heavy metals and hazardous substances, endangers ecosystems and human health. Leachate treatment, recycling, and eco-technological developments are essential for mitigating pollution and progressing towards circular economy models. The following chapter focuses on the methodologies followed in this study.

CHAPTER 3

METHODOLOGY

3.1 Introduction

In this chapter, it is therefore vital to include a description of how the study was done as well as the processes followed to ensure that the study is successfully completed. This chapter provides a clear description of the methodologies followed when conducting data collection and analysis. It provides a broad explanation of the research design, study area, data collection and sampling, research approach, sampling method, analysis of the data and assimilation of the data. These factors were influenced by the study's objective. This chapter makes heavy use of research-focused literature to familiarise the reader with research methodologies and their application.

3.2 Study Area

The study was conducted in Ga-Rankuwa township, which is situated 37 km from Tshwane (the capital city of South Africa) (Figure 3.1). Ga-Rankuwa township experiences hot and rainy summers, and milder fall and winters due to its location, which is between two parts of Pretoria (Pretoria North and Pretoria West) and the neighbouring province of North-West (Maserole & Kgari, 2013). The population of Ga-Rankuwa is approximately 90,945 people with a total of 28,147 households and covers an area of 1,742.83 per km² (Census, 2011). About 99% of the residents are Black Africans, with the majority (69%) speaking Setswana as their first language, 4% English speaking, 4% Xitsonga speaking, 8% Sepedi speaking and 4% Zulu speaking, and a dominating age group of 20 to 24 years followed by 25 to 29 years (Census, 2011).

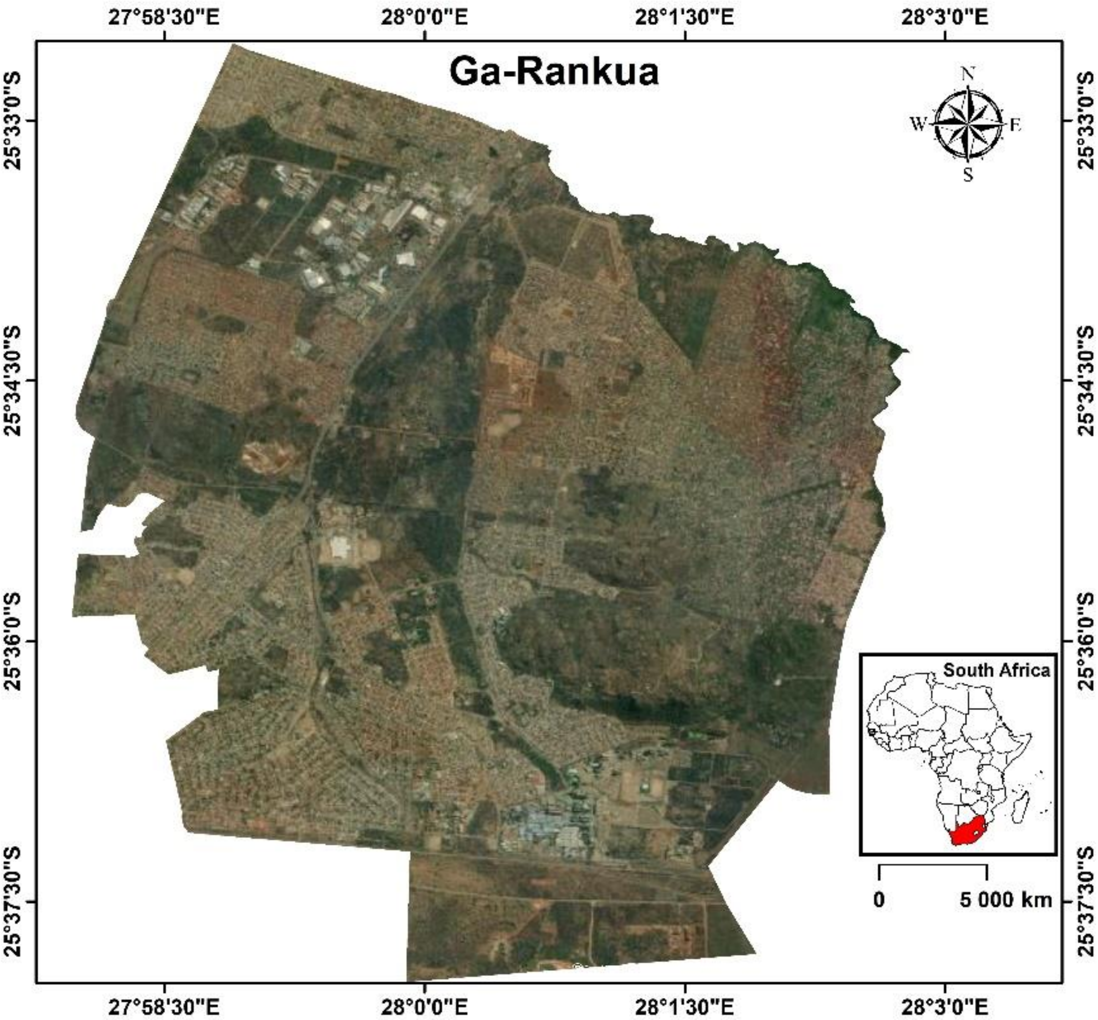

Figure 3.1: Map of Ga-Rankuwa

Provincially, the township currently belongs to the Gauteng province, although under the apartheid regime it was part of Bophuthatswana up until the early 2000s, when it was part of the North-West province (SOHO, 2019). The area is underlain by dolomitic formations of the Transvaal Supergroup, which then increases vulnerability to contamination of groundwater. It receives an average of 603 mm of rainfall per annum, with a daily variation temperature ranging from 5 to 29 °C.

3.3 Research Design

A research design is a map of the research planned by the researcher on how the data will be collected, analysed, and translated into valuable information (Jasti & Kodali, 2012). In this study, a quantitative research approach and a quasi-experiment was used, because the samples were collected as a control to compare the soil and water quality in the landfill area to that in the nearby neighborhood (Asenahabi, 2019). An empirical study was utilised; this is a review which derives and analyses information from immediate or circuitous perceptions (Jasti & Kodali, 2012).

3.4 Safety Measures

Protection from organisms and waste that induce allergic or other severe reactions, such as plants, bees, snakes, hazardous chemical containers and ticks, is a crucial aspect of landfill workplace safety. To minimise cross-contamination during data collection, the researcher used clean gloves when collecting samples and protective gear (including work suits-generally known as overalls and safety boots/shoes that cover the entire foot) when going to the site. Data collection containers and plastics were appropriately labelled beforehand to ascertain that all contents are contained in the relevant container. The area is big enough to move around, and the researcher was accompanied by the study's supervisor during data collection.

3.5 Sampling Procedures

The study employed rigorous field and laboratory protocols to ensure the precision and reliability of environmental assessments. Water samples were systematically collected from three sections of the river upstream, midstream, and downstream, using sterilised, pre-labelled 500 ml containers and standardized contamination prevention measures. Wear Check Laboratory provided the containers to collect water samples. Before sample collection, containers were left unopened and when opened, they were rinsed thrice with sampled water and discarded before a final sample was collected. Samples were stored in a cooler box with ice and transported within 24 hours to the Wear Check Laboratory in Johannesburg for physicochemical and microbial analysis. Upon arrival at Wear Check Laboratory, water samples were refrigerated at 4 °C and analysed within recommended holding times in accordance with standard methods (APHA, 2017). Physiochemical parameters such as pH and electrical conductivity (EC) were measured using calibrated probes. For heavy, metal analysis, water samples were acidified with nitric acid (HNO_3) and analysed using atomic absorption spectrophotometry (AAS). Similarly, soil samples were obtained from 12 ample locations around the landfill at depths of 30–60 cm to reflect potential agricultural exposure zones. The soils were air-dried, sieved through a 2 mm sieve, and stored in carton boxes provided by the Fertiliser Advisory Service Laboratory. The samples were analysed for heavy metals such as zinc, manganese, and copper using acid digestion method and quantification was done using AAS. Microbial activity of water samples was conducted using standard plate count method to determine microbial load. Adherence to established transport and storage protocols ensured sample integrity throughout (Paetz & Wilke, 2015). Data analysis, conducted using XLSTAT, enabled a robust spatial and statistical interpretation of contamination patterns. Collectively, these procedures provided a scientifically sound framework for evaluating the ecological and health implications of landfill-associated pollution.

3.6 Data Collection and Analysis

3.6.1 Water samples

Water samples were randomly collected by directly dipping the containers in the stream to collect water samples. These bottles were only opened when collecting samples to avoid contamination. The collection containers were labelled with section of the river, date and time before use. After sample collection, they were stored in a cooler box with ice and before transportation to the Wear Check laboratory in Johannesburg. Samples were collected in 3 replicates from 3 sections of the river being upstream, midstream and downstream. Before water is collected, the bottles and caps were rinsed three times with the river water and discarded before a final sample was collected. When collecting the sample, the researcher faced the current flow and time was allowed for sediments to settle from the steps if samples were being taken inside the river. Water samples were then transported to the Wear Check laboratory for analysis [(pH), Sodium (Na^{+}), Potassium (K^{+}), Calcium (Ca^{2+}) and Phosphorus (P)]. The samples were stored in ice in a cooler box to avoid any possible microbial activities and were transported to the laboratory within 24 hours of collection. The water samples were also tested for microbial markers such as *Escherichia coli, Listeria monocytogenes, total coliforms* and *Salmonella spp*. Among the many pathogens that may be found in water sources, E. coli is often employed to detect faecal contamination (Ahmed *et al*., 2019; Ali *et al*., 2025). The presence of *Salmonella spp*. and *Listeria monocytogenes* is often seen in agricultural and environmental settings, and these bacteria act as warning signs of potential health concerns and threats to the general population (Chlebicz & Śliżewska, 2018; Williams *et al*., 2023).

3.6.2 Soil samples

Since the landfill is placed in a gentle slope area, the soil sampling provided accurate data as the landscape is not rugged. Three samples on designated sites on the landfill area were taken to provide a more accurate reading. A total of 12 soil samples were conveniently collected in an incremental distance manner following the 4 cardinal points of direction using a soil auger. The samples were collected from a depth of 30 – 60 cm with a spatial separation of 1 km per point. Since home gardening is practised widely nowadays in both rural and urban settings, the 30-60 cm depth is selected due to the rooting depth of certain food crops in order to assess if the potential crops might be affected by the landfill. The heavy metals which were tested are Zinc (Zn), Manganese (Mn), and Copper (Cu) per sample, leading to a total of 36 samples (12 samples x 3 tests each). After collecting, the soils were air dried until completely dry and were sieved through a 2 mm sieve to remove larger particles and other debris from the landfill. Additionally, samples were put into carton boxes provided by the laboratory before being transferred to the Fertilizer Advisory Service labs in Pietermaritzburg for heavy metal analysis. According to Paetz and Wilke (2015), soils should be kept in a cool and dark place during transportation and storage. The results were analysed using XLSTAT which is an add-on software on the Microsoft Excel Package and results presented in form of graphs and tables. In addition to heavy metals, the soil samples were also tested for possible microbial activities.

3.7 Analytical Techniques

Quantitative approaches are approaches that offer decision makers with a systematic and effective method of quantitative data analysis (Ghosh *et al*. 2017). The quantitative approach is a scientific approach used by management for issue solving and decision making. For the purpose of evaluating health risks and water quality, it is necessary to create a number of heavy metal and water quality indices models, such as the degree of contamination, heavy metal evaluation index, contamination factor, other water assessment indices (Withanachchi *et al*., 2018; Bodrud-Doza *et al*., 2016) as well as Heavy-metal Pollution Index (HPI). To categorise soil quality, this approach took into account the maximum allowable levels of each heavy metal in the sampled soils (Mostafa & Islam, 2021). There are many different indices and approaches that may be used in the process of assessing the degree of metal pollution that is present in soils. These approaches and indices are utilised to conduct an analysis of the heavy metals that are already present in the soil. The geo-accumulation index, the contamination factor, and the modified degree of contamination were used for the soil quality assessment objective of this inquiry in order to facilitate the process of determining the amounts of pollution that were present. For biological markers, laboratory results were compared to national and international standards.

3.7.1 Water quality analysis

For analysis of the water data, the following water quality indices were used:

- Trophic State Index (TSI)

The TSI quantifies the degree of eutrophication using Chlorophyll-a concentration (Chl-a), Secchi Disk transparency (SD), and total phosphorus (TP) (Carlson, 1977). This is a scale for classifying nutrient richness and overall quality of the sampled river based on its phosphorus levels. Equation 3.1 was used.

$$TSI_p = 14.42 \times In\,(P) + 4.15 \qquad \text{Eq 3.1}$$

- Where: 14.42 is the scaling factor for determining change rate of the TSI, In(P) is the natural logarithm of the phosphorus content and 4.15 is the intercept value that adjust the TSI score. Water Quality Index (WQI)

The WQI provides an overall measure of water quality by integrating multiple physicochemical parameters (Tyagi *et al.* 2020). Equations 3.2 to 3.4 were used to calculate the WQI.

$$WQI = \frac{\Sigma Q_n \times W_n}{\Sigma W_n} \qquad \text{Eq 3.2}$$

Where: Q_n is quality rating of parameter, and W_n is the unit weight of parameter.

$$Q_n = \frac{C_n}{S_n} \times 100 \qquad \text{Eq 3.3}$$

Where: C_n is the measured value of parameter, and S_n is the standard guideline of parameter.

$$W_n = \frac{1}{S_n} \qquad \text{Eq 3.4}$$

Where: S_n is the standard limit of the parameter. Table 3.2 highlights the limit used in calculating the WQI.

Table 3.1: Acceptable water quality limits

Macronutrient	DWAF (1996) limits
Calcium	80mg/L
Potassium	12mg/L (aesthetic)
Phosphorus	1mg/L (eutrophication)
Sodium	100mg/L

Source: DWAF (1996)

- Nutrient Pollution Index (NPI)

The NPI was used to evaluate nutrient enrichment from nitrogen and phosphorus compounds (Wang *et al*. 2023).

$$NPI = \frac{C_n}{C_s} \qquad \text{Eq 3.5}$$

Where C_n is the measured concentration of nutrient and C_s is the standard concentration for that nutrient (DWAF 1996)

- Percent Sodium (%N)

Percent sodium assesses irrigation water suitability (Ayers & Westcot, 1994). All ionic concentrations are expressed in milliequivalents per liter (meq/L). Values above 60% may adversely affect soil permeability and crop growth.

$$\%Na = \frac{Na^{+} + K^{+}}{Na^{+} + K^{+} + Ca^{2+}} X\ 100 \qquad \text{Eq 3.6}$$

Where: Na^{+} is Sodium, K^{+} is Potassium and Ca^{2+} is calcium

i. Converted mg/L to meq/L using below formula:

$$meq/L = \frac{mg/L}{Equivalent\ weight} \qquad \text{E 3.7}$$

ii. Calculated equivalent weight using below formula:

$$Equivalent\ weight = \frac{Atomic\ weight}{Valence} \qquad \text{E 3.8}$$

- Composite Index (CI)

A composite water quality index integrates individual sub-indices to provide a unified interpretation of water quality (DWAF, 1996). Equation 3.9 was used to calculate CI.

$$CI = \frac{1}{n}\sum_{i=1}^{n}\frac{C_i}{S_i} \qquad \text{Eq 3.9}$$

Where CI is composite index, n is number of parameters, C_i is concentration of parameter, S_i is standard permissible limit. According to the CI ranges, CI<1 = parameters are within limits, CI=1 highlights that parameters exactly meet the standard limits and CI>1 = indicates that the parameters exceed the permissible limits and are a potential for concern.

3.7.2 Soil quality analysis

For analysis of the data, the following soil quality indices and statistical analysis were used:

- Soil Quality Indices
  - Geo-accumulation Index (Igeo)

The Igeo was applied to evaluate the extent of heavy metal accumulation in soils relative to background concentrations. It was computed using Equation 3.10 as formulated by Müller (1969), which incorporates a logarithmic scale to classify soils as uncontaminated, moderately contaminated, or heavily contaminated based on metal enrichment levels.

$$Igeo = \log 2 \left(\frac{\mathrm{Cn}}{1.5 \text{ x } \mathrm{Bn}}\right) \qquad \text{Eq 3.10}$$

where Cn is the measured metal concentration and Bn is the background concentration. Soils are classified from uncontaminated (Igeo ≤ 0) to extremely contaminated (Igeo > 5)

  - Contamination Factor (CF)

The CF was determined for each metal to estimate the contamination intensity by comparing observed concentrations with natural background values. CF values were then used to interpret contamination severity according to Hakanson’s (1980) classification criteria. Equation 3.11 was used to calculate CF.

$$CF = \frac{\mathrm{Cmetal}}{\mathrm{Cbackground}} \qquad \text{Eq 3.11}$$

Values CF<1 indicate low contamination; 1≥CF<3 moderate; 3≥CF≤6 considerable; and CF>6 very high contamination.

  - Modified Degree of Contamination (mCd)

The mCd was calculated to provide an integrated measure of overall soil contamination across multiple metals. It represents the average of individual CF values and serves as a useful index for comparing contamination levels among sites (Abrahim & Parker 2008). Equation 3.12 was used to calculate mCd.

$$mCd = \frac{\Sigma \mathrm{CFi}}{\mathrm{n}} \qquad \text{Eq 3.12}$$

Where n is the number of analysed metals. Higher mCd values denote more severe cumulative pollution.

- Statistical analysis
  - Pearson’s Correlation

Pearson’s correlation (r) was used to quantify linear relationships between physicochemical parameters and heavy metals, identifying possible shared sources or geochemical associations.

  - Analysis of Variance (ANOVA)

ANOVA was used to test for significant spatial differences in parameter values across sites and distances ($p < 0.05$ indicating significance).

  - Multiple Linear Regression (MLR)

MLR models were used to predict water and soil quality indices based on independent variables such as pH, electrical conductivity, and nutrient levels, clarifying dominant predictors of contamination.

All analyses were performed using Microsoft Excel and IBM SPSS Statistics (Version 28). This integrated use of indices and statistical tools provided a comprehensive evaluation of the landfill’s environmental influence on the surrounding soil and water.

3.8 Chapter Summary

This chapter examined research methodology, study strategies, and quantitative data gathering methods. The chosen methods of data collection were acceptable for the region of investigation since a great deal of information regarding the impact of landfills on soil and water quality and the resulting problems was detected. The information obtained will be utilised to improve the situation of Ga-Rankuwa regarding its landfill management difficulties. The subsequent chapter 4 restates the study's goals and provides an analysis the soil and water sample findings. The chapter helps the researcher to determine if environmental ordinances have been appropriately drafted and executed.

CHAPTER 4

RESULTS AND DISCUSSIONS

4.1 Introduction

This chapter summarises, interprets, and discusses the findings as guided by both inductive and deductive techniques, in order to address the research objectives and contribute to the body of knowledge established by this study. The chapter's results are substantial and were based on a large amount of quantitative study material that the researcher analysed and interpreted (Hesse-Biber & Leavy 2011). The researcher directed the research findings (this chapter) based on the promises given in Chapter One (research objectives driving the study), the literature as outlined in Chapter Two, and the data analysis presentation approach as outlined in the next section. The following discussions were designed to clarify and support the study's findings.

4.2 Evaluating the Presence of Heavy Metals in Soils Adjacent to the Landfill Site

Landfill sites are significant sources of environmental pollution, particularly with respect to heavy metal contamination. As leachate migrates from landfills into the surrounding environment, it has the potential to infiltrate the soil and alter its physicochemical properties. This poses severe ecological and health risks due to the accumulation of toxic elements like Cadmium (Cd), Lead (Pb), Copper (Cu), Zinc (Zn), Nickel (Ni), and Chromium (Cr), which do not degrade and can persist in the environment for extended periods (Naeem *et al.,* 2025). The evaluation of heavy metals in soils adjacent to landfill sites is thus an essential environmental monitoring objective. It allows for the assessment of the extent of contamination, helps identify pollution trends with respect to distance and depth, and supports the formulation of effective remediation strategies. According to studies conducted by Kekelidze *et al*. (2022) and Sahragard *et al*. (2024), heavy metal concentrations in soils decrease as one move further away from landfills. This indicates that there may be a direct relationship between landfill leachate and soil pollution in specific regions. Evaluating heavy metal concentrations at landfill regions on a regular and systematic basis is essential for sustainable environmental management.

4.2.1 Spatial variability of soil pH adjacent to the landfill

Soil pH is important in assessing the potential availability of valuable nutrients to plants as well as the toxicity of excess levels of essential metals and heavy metals in plants. pH controls nearly all chemical processes in the soil, including hydrolysis, reduction/oxidation, dissolution/ precipitation and adsorption (Prakash *et al.,* 2025). Generally, metals can occur in various forms in soil, with different interstitial forces keeping them in binding to the soil particles. These interactions are very important because the toxicity of metals and other chemical substances depend on the form in which they exist in the environment (Dube *et al.,* 2001; Caporale & Violante, 2016). In this study, soil samples were collected from various points around the landfill site to determine the pH (measured in KCl) and assess potential spatial variation associated with landfill proximity and leachate influence. Figure 4.1 below highlights the pH values of soils in different sample sites.

Figure 4.1 shows that the soil pH ranged from 4.770 at West 1 km to 7.450 at East 1 km, suggesting a slightly acidic to slightly alkaline composition. According to the USDA National Agricultural Research Centres System (2014), soils that fall within the pH range of 6.6 to 7.3 are considered neutral to mildly alkaline, while those below 5.5 are considered very acidic. According to Kicińska *et al*. (2022), heavy metal mobility in soil is greatly affected by changes in soil pH. Typically, metal mobility decreases as pH rises towards alkalinity.

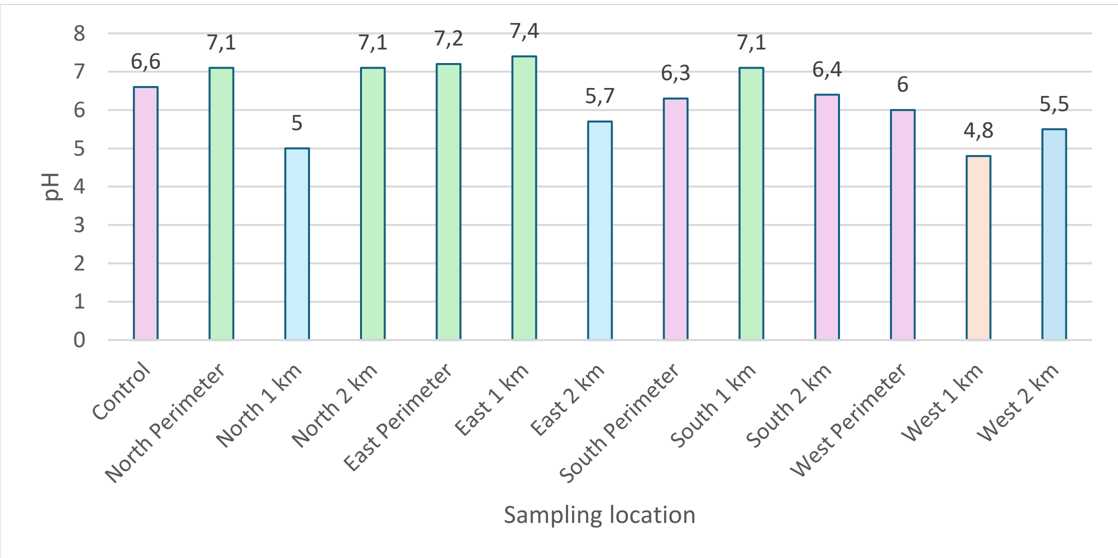

Figure 4.1: The pH values at across all sampling sites

Soils to the west and immediate north of the landfill exhibited more acidic conditions, according to the results of the comparison. As an example, additional studies conducted near landfills have shown acidification because North 1 km had a pH of 5.040 and West 2 km had a pH of 5.520 (Naeem *et al.*

2025; Kekelidze *et al*. 2022). Acidic conditions in such contexts are often associated with elevated concentrations of soluble metals such as Al^{3+} and Mn^{2+}, which can be toxic to plants and inhibit microbial processes (Kabata-Pendias & Mukherjee 2007). Heavy metals under acidic conditions become extremely mobile, resulting in it being available for uptake. In contrast, eastern sites generally exhibited more alkaline pH values. East 1 km (7.450) and East Perimeter (7.170) maintained higher pH levels, possibly due to geological substrate differences or lower exposure to landfill effluents. Alkaline soils can limit the availability of micronutrients like iron, zinc, and manganese (Alloway 2013), potentially impacting plant development if corrective measures such as acidifying amendments are not employed.

At the control site, the pH was 6.520, suggesting a neutral to slightly acidic background. Sahragard *et al*. (2024) state that human activity is likely to blame if the pH level in areas around landfills deviates more than one unit from the baseline. Acidic soils increase metal mobility and bioavailability, which in turn increases ecological and health hazards, as soil pH is a critical factor in determining the solubility of heavy metals (Beinabaj *et al*., 2023; Li *et al*., 2022). Lime supplements may be necessary to raise pH levels and stabilise metal solubility in very acidic soils, such as those at West 1 km (pH 4.770). Alkaline conditions at East 1 km may necessitate the addition of sulphur or organic materials to the soil in order to raise the pH and make micronutrients more accessible (Harter & Naidu 2001). It has also been observed that the patterns of geographical leachate dispersion are associated with the variations in soil pH around landfill sites. Vertical and lateral pH gradients that are in agreement with the paths of leachate movement were also observed by studies conducted by Bongoua-Devisme *et al*. (2018) and Liu *et al*. (2019). The results highlight the need of soil and leachate monitoring techniques that work together.

4.2.2 Presence of heavy metals in soil adjacent to the landfill

The first objective aimed to assess the contamination level of soil with heavy metals. Common heavy metals that were analysed in the soil samples included Copper (Cu), Zinc (Zn) and Manganese (Mn). Table 4.1 analysed the trends and potential sources of contamination in these metals. The Zn concentrations vary significantly across the sampling sites, from as low as 0.248 mg/kg (West 2km) to as high as 56.604 mg/kg (East Perimeter). The East Perimeter stands out with an extremely high value of 56.604 mg/kg, which is far above typical natural concentrations of Zn in soils (usually between 10.00 and 200.00 mg/kg in uncontaminated soils). Metal refining, manufacturing, or agricultural practices, such as the use of Zn-based fertilisers, might all contribute to environmental contamination.

Experimental evidence suggests that soil Zn concentrations more than 300 mg/kg may have detrimental effects on aquatic and plant life (Clemente *et al*., 2003).

The East Perimeter value is likely a result of localised pollution, which could be related to nearby anthropogenic activities, such as mining or industrial effluents. Zinc (Zn) toxicity can cause stunted growth in plants, reduce microbial activity, and contaminate water sources. Furthermore, the South Perimeter had 1.714 mg/kg of zinc (Zn) while the West Perimeter had 1.791 mg/kg of zinc (Zn). Although these values are below the acceptable limits set by regulatory standards for soil quality, their quantity is nonetheless environmentally significant. Increased Zn concentrations, despite adherence to regulatory thresholds, may suggest localised geochemical enrichment or human contributions, such as air deposition from proximate industrial operations, vehicle emissions, or leaching from galvanised substances. The prolonged buildup of Zn, especially in the topsoil, may modify soil microbial populations, influence the nutrient cycle, and, under certain physicochemical circumstances, disperse into adjacent ecosystems. Consequently, although the amounts observed at these sites currently do not offer an immediate contamination threat, ongoing monitoring is essential to identify any possible rising trends that may eventually reach or surpass environmental safety limits. The minimum documented Zn concentrations are found in North 1 km (0.377 mg/kg), West 1 km (0.286 mg/kg), and West 2 km (0.248 mg/kg). These findings approximate natural background levels for Zn, indicating that these places are largely unpolluted by Zn. Table 4.1 below presents the heavy metal distribution in soil samples

Table 4.1: Heavy metals distribution in soil samples (Zn, Mn, Cu)

Sample	Zn (mg/kg)	Mn (mg/kg)	Cu (mg/kg)
Control	1,328	5,469	1,797
North Perimeter	4,531	17,188	9,922
North 1km	0,377	55,660	4,906
North 2km	1,220	7,317	5,203
East Perimeter	56,604	21,698	20,755
East 1km	1,654	11,811	1,890
East 2km	1,589	14,019	2,150
South Perimeter	1,714	238,095	16,476
South 1km	13,279	7,377	6,967
South 2km	1,681	15,929	7,699
West perimeter	1,791	52,239	32,090
West 1km	0,286	2,857	10,095
West 2km	0,248	4,132	4,628

The concentrations of Manganese (Mn) vary from 2.857 mg/kg in the West 1 km to 238.095 mg/kg in the South Perimeter. The South Perimeter has the highest value at 238.095 mg/kg, and the North 1 km follows with the second-highest value at 55.660 mg/kg. Typical natural soil concentrations of manganese are in the range of 10-500 mg/kg; however, the South Perimeter has an extremely high figure of 238.095 mg/kg. This could be due to manmade activity, such as improper waste disposal or pollution, or it might be a consequence of naturally occurring mineral deposits. Soils containing more than 500 mg/kg of manganese may be hazardous, causing plants to grow more slowly and with poorly developed roots (Ducic & Polle 2005). South Perimeter's high Mn levels raise concerns about possible Mn toxicity, which in turn might have an adverse effect on soil microbiome diversity and plant development. Specifically, North 1 km, West Perimeter and East 1 km had elevated Mn concentrations (55.660 mg/kg, 52.239 mg/kg and 21.698 mg/kg, respectively), suggesting that these sites have soil Mn levels that are too high and may be harmful. The sampled areas with lower Mn concentrations include West 1 km (2.857 mg/kg), West 2 km (4.132 mg/kg), control (5.469 mg/kg), North 2 km (7.317

mg/kg) and South 1 km (7.377 mg/kg). There is a variation in Mn concentrations from the perimeter and outwardly, there is no consistent pattern in the concentrations.

There is a considerable difference in copper levels among samples, ranging from 1.890 mg/kg (West 2 km) to 32.090 mg/kg (West Perimeter). At 32.090 mg/kg, the West Perimeter site had the highest Cu values, far higher than the natural background levels (10-100 mg/kg). The applications of Cu-based fungicides and insecticides in agriculture or industrial processes, such as copper mining and metal refining, may be the source of these increased levels. When copper concentrations exceed 100 mg/kg, it may kill plants and soil organisms, causing symptoms including chlorosis, root damage, and decreased microbial diversity (Alloway, 2013). These problems may be exacerbated by the elevated Cu levels in West Perimeter (32.090 mg/kg), which calls for more research into the contamination's origin. Sampled areas that are least severely affected by copper pollution are East 1 km (1.890 mg/kg) and East 2 km (2.150 mg/kg), which have comparatively low Cu levels.

Excessive concentrations of Zn, Mn, and Cu may jeopardise soil health and plant growth. In regions with considerable pollution, elevated Cu levels may disturb soil microbial communities and plant nutrient uptake, whereas high Zn concentrations may reduce microbial diversity, and excessive Mn can induce plant toxicity (Ducic & Polle, 2005). Chlorosis and stunted development are possible outcomes. The toxic effects of Manganese (Mn) on plants have been shown to cause poor root development (Clemente *et al*., 2003; Ducic & Polle, 2005). In the absence of remediation, these heavy metals may contaminate local fauna and make their way into the food chain through crops. Many nations have implemented regulations to prevent soil contamination by heavy metals because of the harm they do to both humans and the environment. Certain areas, especially along the southern and eastern boundaries, may have concentrations of Zn and Mn that are too high, according to this study's results. These outliers highlight the need for ongoing environmental monitoring and focused remedial efforts to reduce ecological and public health hazards. Soil remediation or phytoremediation methods may be necessary to decrease metal levels and restore soil health in areas with very high concentrations of metals, such as the East perimeter (Zn and Cu) and the South perimeter (Mn and Cu).

4.2.3 Analysis of Average Heavy Metal Concentrations at Various Sites

These heavy metals are distributed spatially in soils; Table 4.2 shows the mean concentrations of Zinc (Zn), Manganese (Mn), and Copper (Cu) at five separate sites: Control, Northern site, Eastern site, Southern site, and Western site. This information is critical for environmental monitoring and ecological health. All of these sites demonstrate different degrees of human impact, with the Control site acting as a reference baseline for the study (Table 4.2). The concentration of Zn at the Control site is 1.328 mg/kg, which is very low and falls within the natural background range for uncontaminated soils (10-100 mg/kg). This supports its validity as a benchmark for comparison. the Eastern site exhibits the highest Zn concentration (59.847 mg/kg), substantially exceeding the levels found at all other sites. According to Li *et al*. (2014), this high figure is indicative of pollution in a specific area, which might be caused by emissions from industries and waste disposal. Although the reported level is below the 300 mg/kg threshold at which phytotoxicity begins to occur, it is nevertheless a reason for worry owing to the possibility of bioaccumulation and long-term ecological effects (Nagajyoti *et a*l., 2010; Clemente *et al*., 2003). Additionally, the Southern site had higher Zn contents (16.674 mg/kg) than the Control (1.328 mg/kg), indicating considerable pollution that might be caused by diffuse industrial sources or atmospheric deposition (Wuana & Okieimen, 2011). Zinc concentrations at the Northern and Western sites are 7.128 mg/kg and 2.325 mg/kg, respectively. These results are quite close to the anticipated background levels, suggesting that there is no human effect in these regions.

Table 4.2: Average concentrations (mg/kg) of heavy metals at sites

Sample	Zn (mg/kg)	Mn (mg/kg)	Cu (mg/kg)
Control	1.328	5.469	1.797
Northern site	7.128	80.165	20.031
Eastern site	59.847	47.528	24.795
Southern site	16.674	261.401	31.142
Western site	2.325	59.228	46.813

According to Kabata-Pendias (2011), the average background range for Mn in soils is between 10.00 and 500.00 mg/kg, the southern site had the highest Mn content at 261.401 mg/kg which is a call for concern. Anthropogenic pollution or naturally occurring mineral formations rich in manganese might explain such a high value (Clemente *et al*., 2003). The Mn content at the Northern location was 80.165 mg/kg, lower than the Southern site (261.401 mg/kg), which is still above the typical background level for uncontaminated soils. At the Eastern location, the Mn level was moderate at 47.528 mg/kg, while at the Western site the value stood at 59.228 mg/kg. Although these amounts are lower than those in the Southern and Northern sites, they nonetheless indicate contamination. The contamination might be caused by agricultural runoff or air deposition, which are dispersed sources (Nagajyoti *et al.,* 2010).

The Western site had the highest concentration of Copper (Cu) at 46.813 mg/kg, which is within the normal range for Cu concentrations in soils (10-100 mg/kg) and still causes environmental concern because Cu is highly toxic to soil microorganisms at high levels (Wuana & Okieimen 2011). The second significantly high Cu level (31.142 mg/kg) in the Southern site may have been caused by industrial operations, mining wastes, or agricultural fungicides. Compared to the Western and Southern sites, the Northern (20.031 mg/kg) and Eastern (24.795 mg/kg) sites showed somewhat lower Cu levels, though still suggesting possible contamination, but to a lesser extent. To provide a solid reference point, the Control site had the lowest Cu content (1.797 mg/kg), which is in accordance with soils that are not affected (Kabata-Pendias 2011).

4.3 Analysis of Heavy Metal Concentrations in Soil Samples (Zn, Mn, Cu) using Geo Accumulation Index

According to Gworek (2016), the minimal pollution at the Łubna landfill in Poland revealed that the quantities of heavy metals in the soil were comparable to geochemical background values. One example is the observation of elevated concentrations of lead and cadmium in surface soils (depths of 0-10 cm) close to landfills. Because of bioaccumulation in crops and water systems, these levels often exceed permissible limits, which usually have detrimental consequences on ecosystem health and food safety (Sahragard *et al*. 2024; Naeem *et al*. 2025). Several indicators and techniques may be used to assess soil heavy metal pollution. The levels of pollution were evaluated for this investigation using the geo-accumulation index, contamination factor, and modified degree of contamination formula. The background values utilised were those of Herselman (2007).

4.3.1 Geo-accumulation index

According to Muller (1969), one way to estimate the contamination or enrichment of metal concentrations in soil is to calculate the Geo-accumulation Index (Igeo). This approach measures the level of metal contamination using seven categories, from completely uncontaminated to highly polluted. This research quantifies anthropogenic contamination by computing the Igeo according to Muller's (1969) equation. According to Tibane and Mamba (2022), the amounts of copper (45.00 mg/kg), manganese (850.00 mg/kg), and zinc (95.00 mg/kg) in the background have previously been established.

The values of the Zinc (Zn) Geo-accumulation Index (Igeo) at different sample sites range from -9.17 (West 2 km) to -1.33 (East Perimeter), according to the data. The sites are all classified as "uncontaminated" with an Igeo rating of 0 according to Table 4.3. Corresponding to the comparison, the East Perimeter had the greatest levels of pollution, with a Cn of 56.604 mg/kg and the least negative Igeo value (Igeo = -1.33), suggesting a considerably greater zinc content than the other sites. But it's still in the "Uncontaminated" category, and the levels of contamination from these metals are likely to grow rapidly due to landfill pollution.

Table 4.3: Geo-accumulation index (Igeo) of Zinc (Zn)

Zinc (Zn) (mg/kg)				
Sample sites	**Cn**	**Igeo Value**	**Igeo Class**	**Geo accumulation index (Pollution Level)**
Control	1,328	-6,745563	0	Uncontaminated
North Perimeter	4,531	-4,974909	0	Uncontaminated
North 1km	0,377	-8,56081	0	Uncontaminated
North 2km	1,22	-6,868514	0	Uncontaminated
East Perimeter	56,604	-1,331992	0	Uncontaminated
East 1km	1,654	-6,429257	0	Uncontaminated
East 2km	1,589	-6,486894	0	Uncontaminated
South Perimeter	1,714	-6,377211	0	Uncontaminated
South 1km	13,279	-3,423777	0	Uncontaminated
South 2km	1,681	-6,405141	0	Uncontaminated
West Perimeter	1,791	-6,314017	0	Uncontaminated
West 1km	0,286	-8,962173	0	Uncontaminated
West 2km	0,248	-9,166791	0	Uncontaminated

The West 2 km site had the least amount of pollution, with the lowest Cn value (0.248 mg/kg) and the least Igeo value (Igeo = -9.17). Compared to sites further away from the landfill, the Cn values on the South and East Perimeters tend to be greater, according to the geographical patterns. Because of the materials being placed in the landfill, this would suggest that there are localised sources of zinc contamination near the limits of the research region. There is consistent evidence of lower Cn values at sites West 1 km and West 2 km from the landfill which may indicate that there is minimal zinc deposition away from possible sources and that metals are leaching horizontally rather than vertically, increasing the risk of contamination of groundwater sources. It is well-documented that heavy metals from landfills may reach aquifers by leachate migration along favourable flow routes (Lee *et al*., 2012; Kumar & Alappat, 2005), hence this process raises concerns about the possibility of groundwater contamination.

Although soil zinc levels are well below pollution criteria, it is important to note that these results are based on sampling sites at a certain depth, thus it's feasible that concentrations are greater at lower depths. Consistent with the low indices found in this investigation, Igeo values below zero indicate uncontaminated soils according to Müller's (1969) geoaccumulation index categorisation. Soil Zn contamination at the research sites is unlikely due to the consistently low Igeo classification. In particular, the somewhat higher Igeo values in the East Perimeter compared to the others indicate regional variances in zinc deposition and may need more inquiry into possible local sources of zinc. On the other hand, sites further away from possible pollution sources, such as West 1km and 2 km, East 1 km and 2 km and North 1km and 2 km exhibit very low zinc levels, which is consistent with predictions for such areas. Zhang *et al*. (2019) found that landfills are the most likely causes of zinc fluctuation in the landscape, and this pattern shows that distant places are safer than perimeter zones.

Table 4.4: Geo accumulation index (Igeo) of Manganese (Mn)

Manganese (Mn) (mg/kg)				
Sample sites	**Cn**	**Igeo Value**	**Igeo Class**	**Geo accumulation index (Pollution Level)**
Control	5,469	-7,86507	0	Uncontaminated
North Perimeter	17,188	-6,212994	0	Uncontaminated
North 1km	55,66	-4,517703	0	Uncontaminated
North 2km	7,317	-7,445015	0	Uncontaminated
East Perimeter	21,698	-5,876784	0	Uncontaminated
East 1km	11,811	-6,754219	0	Uncontaminated
East 2km	14,019	-6,507002	0	Uncontaminated
South Perimeter	238,095	-2,420887	0	Uncontaminated
South 1km	7,377	-7,433238	0	Uncontaminated
South 2km	15,929	-6,322679	0	Uncontaminated
West Perimeter	52,239	-4,609232	0	Uncontaminated
West 1km	2,857	-8,801708	0	Uncontaminated
West 2km	4,132	-8,26936	0	Uncontaminated

The Geo-accumulation Index (Igeo) analysis shows that all sample sites have negative Igeo values, which classify them as Class 0 (Uncontaminated) according to Müller's Igeo scale (Müller, 1969). The negative values range from about -2.42 at the South Perimeter site to -8.80 at the West 1 km site. The South Perimeter has the greatest relative manganese enrichment (Igeo ≈ -2.42), which might indicate that it is the area most affected by pollution sources related to landfill leachate migration or surface runoff. Regardless of the relative elevation, the site is still classified as uncontaminated, meaning that the Mn levels are still within the normal background fluctuation. Alternatively, the West 1 km site has the lowest manganese level (Igeo ≈ -8.80), which indicates that there is lower Mn enrichment compared to baseline values. This might be because it is further from the landfill core and has less exposure to routes that contaminants can be transported (Jiang *et al*. 2019). Compared to further areas, such as West 2 km (-8.269) and North 2 km (-7.445), spatial patterns show that sites situated around the landfill boundary, such the East Perimeter (-5.876), North 1 km (-4.517), and West Perimeter (-4.609), tend to display greater Mn concentrations. Metal concentrations normally decrease with increasing distance from the source owing to attenuation processes like sorption, precipitation, and dilution in surrounding soils (Alloway 2013; Wuana & Okieimen 2011), which is consistent with this distribution and contaminant dispersion models related to landfills. Although the manganese levels at all sites are below safe limits, the results show that the South Perimeter and other near-field regions should be monitored for their higher amounts. Landfills need this kind of monitoring because trace metal buildup in nearby soils may rise over time due to protracted leachate migration (Kjeldsen *et al*., 2002).

The Geo-accumulation Index (Igeo) was also used to assess the degree of copper (Cu) contamination in soil samples and classify pollution levels across the study area. Table 4.5 presents the Igeo values for copper, indicating the contamination status at the sampled sites.

Table 4.5: Geo accumulation index (Igeo) of Copper (Cu)

Copper (Cu) (mg/kg)				
Sample sites	**Cn**	**Igeo Value**	**Igeo class**	**Geo accumulation index (Pollution Level)**
Control	1,797	-5,231326	0	Uncontaminated
North Perimeter	9,922	-2,766203	0	Uncontaminated
North 1km	4,906	-3,782368	0	Uncontaminated
North 2km	5,203	-3,697402	0	Uncontaminated
East Perimeter	20,755	-1,701448	0	Uncontaminated
East 1km	1,89	-5,15861	0	Uncontaminated
East 2km	2,15	-4,972793	0	Uncontaminated
South Perimeter	16,476	-2,034505	0	Uncontaminated
South 1km	6,967	-3,276234	0	Uncontaminated
South 2km	7,699	-3,132123	0	Uncontaminated
West Perimeter	32,09	-1,072784	0	Uncontaminated
West 1km	10,095	-2,741213	0	Uncontaminated
West 2km	4,628	-3,866396	0	Uncontaminated

According to table 4.5 of Cu igeo above, every site is classified as "0 (uncontaminated)" due to the negative pollution levels, which range from around -1.07 (West perimeter) to about -5.23 (Control). The West perimeter had the greatest amount of pollution, with a Cu Igeo value of -1.07, the highest of all the sites (Table 4.5). Though it is still considered to be uncontaminated, the drastically higher amounts of the metal in the West perimeter compared to other areas indicate that it is closer to possible sources of copper, such as industrial operations, waste disposal with a high copper content, or even runoff. Furthermore, the concentrations are rather also high in the East perimeter and the South perimeter, with Igeo values of around lgeo≈−1.70 and lgeo≈−2.03, respectively. The Control site had the lowest copper concentration (Cn = 1.797 mg/kg) and the lowest Igeo value (Igeo≈−5.231), indicating the absence of significant pollution. East 2 km (Igeo = -4.972) and East 1 km (Igeo = -5.158), which are also located farther from the perimeters, also exhibit very low copper levels. Soil and sediment concentrations of Cu have been shown to follow distance-decay patterns. This is thought to be due to sorption processes, dilution, and the fact that metals from localised sources are not broadly dispersed (Alloway 2013; Islam *et al.* 2015).

Additionally, the copper concentrations on the East and South Perimeters are also relatively high, with Igeo values of around -1.70 and -2.03, respectively. These areas are more likely to have been subject to moderate human activities and/or the vertical movement and leaching of this metal than less impacted areas. Copper concentrations at the West Perimeter (Igeo=-1.072) compared to the control site (Igeo=-5.2313) suggest localised deposition or human interference in this region. Copper concentrations are often greater in perimeter sites (for example, West, East, and South) compared to outlying sites (for example, 1 km or 2 km distances). This might mean that there is a higher concentration of copper-containing trash close to the research region's borders, because there are more copper-containing sources in that area. In line with predictions for sites less affected by external pollution, control and sites further from the perimeters (for example, North 2 km, West 2 km, East 2 km) display negligible copper levels.

Although not polluted, the West (lgeo = -1.07) and East perimeter (lgeo ≈-1.70) sites stand out with much higher Igeo readings than the others. It is necessary to periodically examine these sites since they may be closer to copper enrichment sources. However, its mobility may be enhanced in leachates that are acidic or rich in organic compounds, which might cause vertical leaching and groundwater pollution (McBride, 1994; Dijkstra *et al*., 2004). The increased copper contents at the West boundary need additional examination, particularly if there are close industrial or agricultural activity, even if all sites come under the "uncontaminated" category. Over time, these actions may increase concentrations.

4.3.2 Comparison of Soil Contaminants (Zn, Mn, Cu) with WHO Permissible Limits

To evaluate possible hazards to human and environmental health, the World Health Organisation (WHO) has set recommended levels for soil heavy metal concentrations (WHO 2005). The permissible limits are shown in Table 4.6. Alloway (2013) states that uncontaminated soils often have background Zn values ranging from 10.000 to 30.000 mg/kg. Between 0.248 mg/kg (West 2 km) to 56.604 mg/kg (East Perimeter), zinc concentrations vary across all sites. Contamination is not a possibility since all levels are far below than the 300 mg/kg limit set by the World Health Organisation. With values ranging from around -9.16 (West 2 km) to about -1.33 (East perimeter), the Igeo values categorise all sites as 0 (uncontaminated). Though plants cannot grow without manganese, soil levels over 850 mg/kg are harmful. Soil manganese pollution is uncommon outside of industrial zones, according to research by Kabata-Pendias (2010). With concentrations ranging from 2.857 mg/kg (West 1 km) to 238.095 mg/kg (South perimeter), the South Perimeter site shows the highest Mn levels, although they are still below acceptable limits. The WHO does not consider these amounts to be harmful since they are all below the usual threshold levels of concern, which are generally 850 mg/kg or more. With values ranging from around -8.80 (West 1 km) to about -2.42 (South boundary), the Igeo values categorise all sites as 0 (uncontaminated).

Table 4.6: WHO permissible limits for heavy metal concentration in soil

Heavy metal	Permissible limits for heavy metals in soil
Zinc (Zn)	300 mg/kg
Manganese (Mn)	500 – 1000 mg/kg
Copper (Cu)	100 mg/kg

Typically, uncontaminated soils will have copper contents ranging from 2-50 mg/kg. Industrial or agricultural operations are often linked to elevated levels. Copper concentrations exceeding 100 mg/kg may be indicative of human-induced causes (Alloway 2013). From the control (1.797 mg/kg) to the west boundary (32.090 mg/kg), the concentrations of Cu vary. All values are below the WHO permissible limit of 100 mg/kg, indicating no significant contamination. Although the results may vary over time, they are consistent with previous studies on soil background metal levels and lend credence to the idea that these sites are unaffected by pollution (Li *et al*., 2021; Boechat *et al*., 2020).

4.4 Contamination Factor and Modified Degree of Contamination Indices

The contamination level of these sample sites was determined by analysing the distribution of heavy metals in soils using the Contamination Factor and the Modified Degree of Contamination indices. To gauge the extent of pollution and identify threats to ecosystems and human health, these indices are useful. The results are presented in Table 4.7.

Table 4.7: Distribution of heavy metals in soils using contamination factor and modified degree of contamination index

Site	Zn (mg/kg)	Mn (mg/kg)	Cu (mg/kg)	CF (Zn)	CF (Mn)	CF (Cu)
Control	1.328	5.469	1.797	0.014	0.006	0.040
North Perimeter	4.531	17.188	9.922	0.048	0.020	0.220
North 1km	0.377	55.660	4.906	0.004	0.065	0.109
North 2km	1.220	7.317	5.203	0.013	0.009	0.116
East Perimeter	56.604	21.698	20.755	0.596	0.026	0.461
East 1km	1.654	11.811	1.890	0.017	0.014	0.042
East 2km	1.589	14.019	2.150	0.017	0.016	0.048
South Perimeter	1.714	238.095	16.476	0.018	0.280	0.366

South 1km	13.279	7.377	6.967	0.140	0.009	0.155
South 2km	1.681	15.929	7.699	0.018	0.019	0.171
West Perimeter	1.791	52.239	32.090	0.019	0.061	0.714
West 1km	0.286	2.857	10.095	0.003	0.003	0.224
West 2km	0.248	4.132	4.628	0.003	0.005	0.103

4.4.1 Contamination Factor

The control site's CF values are very low, as seen in Table 4.7. It reflects the fact that, relative to the baselines, it is basically free from heavy metal pollution, as manganese (Mn) (0.006) < zinc (0.014) < copper (0.040). At the north perimeter, the CF follows the trend Mn (0.020) **<** Zn (0.048) < Cu (0.220), which indicates that there is little pollution, with Cu being the most prevalent metal. North 1 km, North 2 km, East 1 km, and West 1 km sampling sites generally have low contamination (CF values close to 0 or below 1), indicating minimal contamination of Zn, Mn, and Cu. Nonetheless, metal contamination levels range from moderate to high at sites around the East, South, and West Perimeters. For instance, the East Perimeter site shows a modest level of Mn (0.026), but a relatively high CF for Zn (0.595) and Cu (0.461). Also, the pollution levels are low to moderate in the South 1 km Mn (0.009) < Zn (0.140) < Cu (0.155) and West 2 km Zn (0.003) < Mn (0.005) < Cu (0.103) sites. Overall CF values for Cu are high in all sampled sites, with the exception of the CF value for Zn in East Perimeter (0.596). Dijkstra *et al*. (2004) and Lee *et al*. (2012) found that metals disperse horizontally through lateral migration of leachates rather than uniformly into subsoil layers. This suggests that anthropogenic input is reduced as CF values gradually decline with distance from the landfill.

4.4.2 Modified Degree of Contamination

According to Hakanson (1980) and Abrahim & Parker (2008), the modified Contamination degree (mCd) index is a more comprehensive way to assess possible ecological risk than just looking at individual Contamination Factors (CF). It gives a more holistic evaluation of the cumulative contamination risk from multiple metals at each site. According to the widely used mCd classification, where values below 1.5 indicate low overall pollution (Abrahim & Parker 2008), the current study found mCd values ranging from 0.020 at the Control site to a maximum of 0.361 at the East Perimeter (Figure 4.2), suggesting generally low contamination across the study area. The East Perimeter had the highest mCd value (0.361), which was caused by relatively high amounts of copper (2.075 mg/kg, CF=0.461) and zinc (56.604 mg/kg, CF=0.596). Because the landfill is commonly filled in stages and closed, this might insinuate that some of the waste dumped towards the east of the landfill was rich in Cu and Zn. According to Christensen *et al.* (2001) and Kjeldsen *et al*. (2002), areas with high mCd values are more likely to experience contaminant migration from waste material, which might explain why they receive a higher cumulative metal intake than other areas.

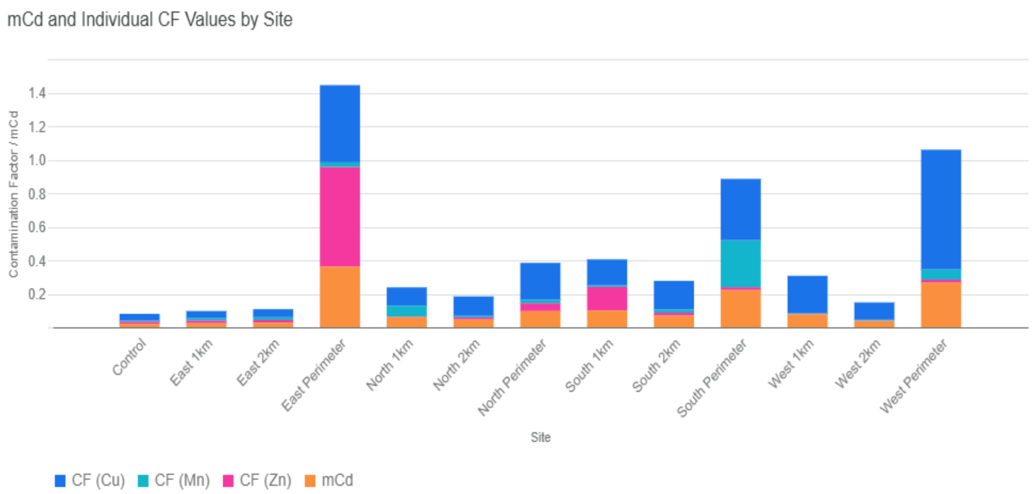

Figure 4.2: Spatial variations of metals driving contamination using mCd (Zn, Cu, Mn)

While still falling into the low contamination category, the West Perimeter and South Perimeter both demonstrated relatively high cumulative contamination, with mCd = 0.265 and mCd = 0.221, respectively. While a high Cu contamination factor (CF=0.714) was a major cause in the higher mCd at the West Perimeter, Manganese (Mn) at an extraordinarily high concentration of 238.095 mg/kg (CF=0.280) was the main contributor at the South Perimeter. As a composite metric, mCd is useful because it takes into consideration the different ways in which different metals contribute to the overall contamination load. Sites such as the North Perimeter had moderate mCd values (0.096) and the South 1 km (0.101), which indicates a little enrichment above background levels for several metals but is still within acceptable environmental limits. East 1 km (0.024), East 2 km (0.027), and West 2 km (0.037) were some of the most distant sites, with the lowest mCd values. This is likely due to the fact that these sites were less exposed to the mechanisms that transport contaminants and that the attenuating effect of distance from the landfill source was at work.

In general, there is a noticeable perimeter effect in the geographical distribution of mCd values; areas closer to adjacent sites had higher contamination degrees, whereas sites further away from the landfill had lower values. The elevated mCd values at the East Perimeter, West Perimeter, and South Perimeter should be monitored further because, over time, leachate migration and metal accumulation could cause these sites to move from the "low contamination" category to a higher contamination risk category (Wuana & Okieimen 2011).

4.5 Potential Impacts of Heavy Metal Contamination

Table 4.7 indicates that Cu exhibits the highest concentration among metals in the soils at 0.7131 mg/kg, whilst Mn has the lowest concentration at 0.0026 mg/kg. This metal may adversely affect the liver, cause anaemia, and impair the intestines and kidneys when present at elevated concentrations in both humans and animals (Leal *et al*., 2023). A variety of anthropogenic activities, including sewage treatment facilities, landfills, fertilisers, agricultural goods, and sewage effluent, may contribute to environmental Cu pollution (DWAF, 1996; Lenntech, 2022). Several health problems, including cancer, liver and kidney damage, gastrointestinal discomfort, and red blood cell dysfunction, have been linked to human exposure to high amounts of Cu (Mahurpawar, 2015). Cu impacts in soils are also associated with decreased plant-water and nutrient uptake, which hinders plant development (Shabbir *et al*. 2020). At the eastern boundary, the soil Zinc concentration was 0.5958 mg/kg, but at the western 2 km mark, it was just 0.0026 mg/kg. Zinc has cellular-level effects on human organs and could play a pivotal role in regulating cell death and neuronal death after brain damage (Plum *et al*., 2010). According to Sharma and Agrawal (2005) and Singh and Kalamdhad (2011), there are other health problems linked to human exposure to high amounts of Zn metal, including kidney and liver failure, worsening Alzheimer's symptoms, blood in urine, and gastrointestinal issues. Soil has very little mobile zinc, making it easy to absorb and store (Lenntech, 2022). On the other hand, aquatic habitats may be at risk from excessive Zn in water since it may raise acidity (Lenntech, 2022). The lowest amount observed for Mn was 0.0034 mg/kg (West 1 km), while the greatest quantity was 0.2801 mg/kg (East 1 km). According to McAllister (2011) and Rollin *et al*. (2005), excessive levels of Manganese in humans may cause harmful toxic consequences, such as damage to the nervous system, anxiety, and sleeplessness. A soil solution containing Mn^{2+} ions is produced when silicates weather, and plants readily acquire these ions (Chibuike & Obiora 2014).

4.6 Statistical Analysis

To examine the metal correlations in the soil samples, three statistical tests were used. Below are the findings that were obtained by the use of linear regression, one-way Analysis of Variance (ANOVA), and Pearson's correlation coefficient (r).

4.6.1 Pearson's correlation

To find the linkage between the soil heavy metal concentrations at various sample sites and their geographical and temporal relationships, Pearson's correlation is required. A table displaying the significant correlation coefficients between soil pH and soil heavy metals is provided in Table 4.8. Correlation coefficients that had a probability level less than 1 ($p < 0.05$) were deemed significant.

Table 4.8: Pearson correlation coefficients (r) among heavy metals in soil samples

	pH (KCL)	Zn (mg/kg)	Mn (mg/kg)	Cu (mg/kg)
pH (KCL)	1			
Zn (mg/kg)	0.383367	1		
Mn (mg/kg)	-0.10295	-0.08532	1	
Cu (mg/kg)	-0.0096	0.371878	0.368412	1

Table 4.8 shows that there is a positive weak correlation between soil pH and Zn (0.0383367), Zn and Cu (0.371878), and Mn and Cu (0.368412) meaning that when one variable increases, the other tends to moderately increase too. This indicates that these metal pairings may have same geochemical routes or come from similar origins within the landfill-impacted environment. One possible explanation for these positive connections is that heavy metals leached into the soil from previously deposited waste. However, there was no meaningful relationship between pH and Cu (-0.0096). On the other hand, there was a moderate negative relationship between pH and Mn (-0.10295) and Zn and Mn (-0.08532), meaning these metals almost have no effect on each other. When one of the metals or metals against pH grew or dropped, the other did not follow the same trend, as shown by these weak associations (Xu *et al*. 2022). These correlations were reached at a 95% confidence level ($p < 0.05$). These results are supported by similar results from research on heavy metal co-distribution in soils (Kabata-Pendias, 2011; Wuana & Okieimen, 2011). Soil adsorption-desorption characteristics, chemical composition, and physical features all have a role in the bioavailability of heavy metal transport (Xu *et al*. 2022).

4.6.2 Multiple linear regression

The dependent variable (pH) strongly correlates positively with the chosen predictor variables (Zn, Mn, and Cu concentrations), according to the study's regression model, with a Multiple R value of 0.890. The model explains a high power of almost 79.2% of the variance in the dependent variable, according to the R^2 value of 0.792. Adjusted R^2 (0.688) indicates a slightly lower explanatory power after controlling for the number of predictors, suggesting some loss of power after accounting for the number of predictors (Table 4.9). the average distance between observed data points and the regression line, as revealed by the standard error (0.0596), shows fairly accurate predictions.

Table 4.9: Multiple linear regression analysis for sampled heavy metals in soil

Regression Statistics	
Multiple R	0.889798756
R Square	0.791741826
Adjusted R Square	0.687612739
Standard Error	0.059342781
Observations	13

Zhao *et al*. (2022) indicated that metal solubility and speciation may be affected by higher pH values, and our findings are in accordance with that idea. Generally, metal mobility is reduced under alkaline environments. From running the model, a negative correlation was shown by the coefficient (-0.00557, p = 0.00340) on the Zn (mg/kg), where higher Zn concentrations might be linked to a lower dependent variable. According to Wu *et al*. (2020), soil biological activity and associated functions may be diminished when Zn levels are elevated since they are harmful to plants and microorganisms. Moreover, the negative effect of the coefficient (-0.000918, p = 0.0158) experienced on Mn (mg/kg) is consistent with earlier results that excessive amounts of this critical element might harm plant physiology and microbial communities (Millaleo *et al*., 2010). However, there was a positive correlation between Cu (mg/kg) shown by the coefficient (0.00707, p = 0.0170). This might be because Cu, when present in moderate amounts, supports enzymatic activity and plant metabolic processes (Broadley *et al*., 2012).

4.6.3 Analysis of Variance

Using an ANOVA test, it was shown that the metals were statistically distinct, were F7.60 = 4.10, p<0.05. The p-value was 0.007 after correction (Table 4.10). Results at the 0.05 level of significance were obtained from a sample of thirteen different soils. Since $F_{cal} > F_{tab}$, the null hypothesis H_0 is rejected at the 5% level of significance. The conclusion drawn revealed that the three heavy metals are significantly different from one another. Analysis of the sources of heavy metal contamination was possibly caused by the leaching and disposal of waste.

Table 4.10: ANOVA analysis for sampled heavy metals in soil

ANOVA	df	SS	MS	F	Significance F
Regression	4	0.107104398	0.026776	7.603465	0.00783842
Residual	8	0.028172525	0.003522		
Total	12	0.135276923			

Well below the traditional significance level of 0.05, the Analysis of Variance (ANOVA) test yielded an F-statistic of 7.603 and a p-value of 0.0078. It is very improbable that the combined effect of pH, Zn, Mn, and Cu on the dependent variable is attributable to random chance, since this result suggests that the whole model is statistically significant. Alloway (2013) and Kabata-Pendias (2021) state that heavy metals and soil pH are two of the most important factors affecting soil health and the mobility of possible contaminants, which is consistent with our findings.

4.7 Assessing Water Quality from the River near the Landfill Site

4.7.1 Assessment of pH level in sampled river sections

Ion exchange, solubility, and desorption are the three main steps in the heavy metal migration and transformation process (Zhang *et al*. 2018). One of the many parameters that might affect the movement and transformation of metals is pH, which has a significant impact on the speciation of heavy metals (Riba *et al*., 2004; Zhang *et al*., 2018). According to Zhang *et al*. (2018), heavy metal migration and distribution may be affected by changes in system pH. The pH of the surrounding environment is thought to be the most important factor affecting the leachability of heavy metals (Dijkstra *et al*., 2004; van der Sloot & Kosson, 2010; Król, 2011; van der Sloot & van Zomeren, 2012; Saveyn *et al*., 2014). The pH values of the river sections that were sampled are summarised in Table 4.11.

Table 4.11: pH level of the sampled water

Sample	pH values
Upper river	7.24
Mid river	7.48
Lower river	7.70

There is a noticeable rise in pH levels from the upper to the lower sections of the river. The upper river segment recorded a pH of 7.24, indicating a slightly alkaline condition. A more markedly alkaline environment was revealed downstream by the lower river segment's pH of 7.70, in contrast to the mid-river site's pH of 7.48 (Table 4.11). Several geochemical and hydrological processes, including dilution effects, carbonate dissolution, and the impact of human discharges, may be responsible for the trend of rising alkalinity along the river gradient. Consistent with previous research by Chapman (1996) and Wetzel (2001) on river systems, this study also reveals that pH values gradually rise from upper river to the lower river as dissolved ions from weathering and wastewater inflows accumulate. Bicarbonates and carbonates, which are often mobilised from nearby geological formations and agricultural runoff, may also have an effect on the alkalinity (Zhou *et al*., 2019). The documented pH levels align with the range advised by the WHO and the USEPA, namely 6.5 to 8.5, for the majority of freshwater aquatic species (WHO, 2022; USEPA, 2018). On the other hand, Bhatnagar and Devi (2013) found that above pH of 7.70, some metal ions may start to precipitate, which might change nutrient dynamics and reduce their bioavailability. The preservation of aquatic biodiversity and the administration of water quality are both affected by this. The study conducted by Slack *et al*. (2005) identified elevated concentrations of heavy metals in a river located downstream of a dump, signifying evident contamination.

4.8 Contamination of the River Quality by Heavy Metals and Macronutrients.

Industrial production and usage, mining and smelting, as well as home and agricultural uses of metals and metal-containing compounds, are the primary causes of environmental pollution (EEA 2000; Wuana & Okieimen, 2011). According to Simeonov *et al*. (2011) and Tchounwou *et al*. (2012), heavy metals may enter the environment via a variety of pathways, including atmospheric deposition, soil erosion, heavy metal leaching, sediment re-suspension, and metal evaporation from water sources into soil and groundwater. Manufacturing facilities for metals, coal, petroleum, nuclear power, textiles, microelectronics, wood preservation, paper, and other industrial products are among the sources (Joshia *et al.* 2016). The degree of pollution of the river, whether for human consumption or for aesthetic reasons, has been evaluated using several criteria. Sodium concentration/percentage, water quality index, nutrient pollution index, and the Trophic State Index (TSI) for Phosphorus are some of the metrics that have been used and examined in this context.

4.8.1 Trophic State Index for Phosphorus

The TSI was calculated by converting the phosphorus levels from mg/L to µg/L through multiplying by 1000, as shown in Table 4.12. Table 4.12 shows that all the computed TSI values are more than

100, putting the circumstances squarely in the hypereutrophic range and suggesting an abundance of nutrients. Severe algal blooms, low dissolved oxygen during decomposition, and possible fish fatalities are common outcomes of such high phosphorus levels (Smith *et al*., 1999; Dodds, 2006). A possible point-source or concentrated runoff effect upstream might be indicated by the greatest phosphorus level (4540 µg/L) and TSI (125.49) in the midriver segment.

Table 4.12: TSI analysis for phosphorus

Location	Phosphorus (mg/L)	Phosphorus (µg/L)	TSI
Upper River	2.21	2210	115.23
Mid River	4.54	4540	125.49
Lower River	1.68	1680	111.14

The lower river has a somewhat reduced TSI (111.14), potentially attributable to dilution or sedimentation effects downstream. Nonetheless, even the minimal reading is ecologically concerning. Increased phosphorus levels are frequently associated with agricultural runoff, the discharge of untreated wastewater, and the erosion of phosphorus-laden soils (Carpenter *et al*., 1998). In river systems, such circumstances expedite eutrophication, deteriorating aquatic ecosystems and elevating water treatment expenses. Restoring riparian buffers and implementing stronger effluent restrictions are two urgent nutrient management treatments that must be implemented immediately to prevent further deterioration.

4.8.2 Water Quality Index assessment for river quality

To get a good picture of the water's quality, it's important to look at the Water Quality Index (WQI) criteria (Nitka *et al*., 2019dc), and the summary of the WQI findings is shown in Table 4.13. The water quality of the upper river is very poor, having a WQI of 207.53. Even if the WQI readings are lower than those in the midriver, they still show an elevated amount of pollution, especially from phosphorus (221 mg/L), which is almost double the allowed limit. The transfer of dissolved and particulate-bound nutrients into nearby bodies of water occurs as a result of landfill leachate's lateral migration through soils and surface runoff pathways, this phenomenon has been extensively documented in river systems close to landfill sites (Lee *et al*., 2012; Naveen *et al*., 2017; Barton *et al*., 2019). Looking at midriver where pollution is most noticeable, the water quality seems to deteriorate even further, with a WQI of 422.32 signifying the poorest possible water quality and making it unfit for human consumption. Contaminants in the midriver have presumably grown as a result of landfill discharge or leachate. The high phosphorus levels are particularly concerning as they may result in eutrophication, which can damage aquatic ecosystems. Eutrophication resulting from phosphorus pollution is a well-documented problem next to landfills and agricultural runoff zones, where high phosphorus induces algal blooms and diminishes oxygen levels in aquatic environments (Smith, 2003). The elevated amounts of potassium, sodium, and phosphorus indicate significant pollution, presumably resulting from the landfill.

Table 4.13: Water Quality Index assessment

Macronutrient			**Upper**		**Mid**		**Lower**	
	Wn	DWAF (1996) limits	Qn	Qn×Wn	Qn	Qn×Wn	Qn	Qn×Wn
Calcium	0.0125	80mg/L	64.25	0.8031	56	0.7	62.38	0.7798
Potassium	0.0833	12mg/L (aesthetic)	86.67	7.2225	140	11.662	75.67	6.3033
Phosphorus	1	1mg/L (eutrophication)	221	221	454	454	168	168
Sodium	0.01	100mg/L	46.6	0.466	64.6	0.646	4.54	0.0454
WQI			**207.53**		**422.32**		**158.37**	

The lower river exhibits a WQI of 158.37, reflecting a notable enhancement in its cleanliness and a decrease in pollution levels. The quality of the water continues to raise concerns; however, natural processes like sedimentation and dilution seem to improve it as the river flows away from the landfill. Nevertheless, the attenuation has not reduced the very high phosphorus levels (168 mg/L), which further indicates that the river system is still at danger of eutrophication. Earlier research by Saeedi *et al*. (2010) and Zubair *et al*. (2016) found similar results in other catchments affected by landfills. In these areas, nutrient loads still affect aquatic systems many kilometres downstream, although at lower concentrations. According to Afolayan and Fagade (2010), there has to be an improvement in waste management in order to protect water quality, and this study shows that the landfill's effects on the river are continuous.

4.3.8 Nutrient Pollution Index

The ratio of a nutrient's measured concentration (C_n) to its standard or allowable limit (C_s) is known as the Nutrient Pollution Index (NPI). The sodium NPI, significantly greater than 1, is the principal factor influencing the overall NPI at each location. Elevated sodium NPI may signify point-source discharge from industrial or household saline effluent, road salt application, or concentrated dissolution of saline substances. The environmental consequences encompass heightened soil sodicity risk (soil dispersion, crusting, diminished infiltration) and possible effects on freshwater biota that are resistant to ionic fluctuations (Qadir *et al.*, 2018; Matheyarasu *et al.*, 2016). Table 4.14 highlights the NPI of the calculated macronutrients.

Table 4.14: Nutrient Pollution Index assessment

Macronutrient	**Standard concentration (DWAF, 1996)**	**Upper**		**Mid**		**Lower**	
	Cs (mg/L)	Cn (mg/L)	NPI= Cn/Cs	Cn (mg/L)	NPI= Cn/Cs	Cn (mg/L)	NPI= Cn/Cs
Calcium	32	51.4	1.61	44.8	1.4	49.9	1.56
Potassium	10	10.4	1.04	16.8	1.68	9.08	0.91
Phosphorus	0.1	46.6	466	64.6	646	45.9	459
Sodium	100	2.21	0.02	4.54	0.05	1.68	0.02
NPI_{ave}		**117.17**		**162.28**		**115.37**	

There is a large range in the computed Nutrient Pollution Index (NPI) values shown in the table, with averages of 117.17 (upper), 162.28 (mid), and 115.37 (lower) for the various sample places. These averages show that, in comparison to the amounts recommended by the Department of Water Affairs and Forestry (DWAF, 1996), the nutrient contamination is worse near the middle of the river. As a result of nutrient enrichment beyond acceptable limits, there is a higher risk of eutrophication and the corresponding degradation of water quality when NPI values are elevated, particularly above 1.0 (Srinivasan *et al.*, 2017). Phosphorus concentrations are very high compared to other macronutrients; NPI values vary between 459 and 646 across sites, which is much higher than the prescribed limits. Trace amounts of phosphorus may have a negative impact on aquatic ecosystems by increasing the frequency and severity of algal blooms, decreasing the amount of oxygen in the water, and generally making freshwater systems less functional (Correll, 1998; Schindler *et al.*, 2016). On the other hand, there is a negligible concern for eutrophication since the sodium concentrations are far lower (NPI = 0.02) than the normal limits (NPI>1). Despite being within safe levels, calcium and potassium still contribute to the total nutritional load since their NPI values are near or slightly above 1. The high mid-river NPI average (162.28) indicates cumulative contributions from human activities, including agricultural runoff and wastewater discharges, aligning with research that emphasises mid-stream nutrient surges linked to land use intensity and effluent inflows (Smith & Schindler, 2009; Jarvie *et al.*, 2013). Due to dilution, sedimentation, or nutrient absorption by aquatic biota downstream, the lower-river segment has a lower NPI average than the mid-part (Dodds & Smith, 2016).

4.8.3 Percent Sodium (PI) assessment

Soil permeability, structure, and crop yield may be negatively impacted by excessive Na levels compared to Ca, P, and K, hence evaluating irrigation water quality generally entails assessing Percent Sodium (%Na). Reduced infiltration, poor aeration, and long-term losses in agricultural output are the results of irrigation water that contains too much salt, which promotes soil dispersion (Ayers & Westcot, 1994; Todd & Mays, 2005). The concentrations of macronutrients in the upper, mid, and lower river sections were used to calculate %Na, which might be used to identify potential challenges for irrigation.

Table 4.15: Percent Sodium assessment

Site	Ca (mg/L)	K (mg/L)	P (mg/L)*	Na (mg/L)	%Na	Irrigation Suitability**
Upper River	64.25	86.67	221	46.60	20.3%	Excellent to Good
Mid River	56.00	140.00	454	64.60	23.9%	Excellent to Good
Lower River	62.38	75.67	168	4.54	3.6%	Excellent

*Phosphorus included for completeness of water quality analysis but not used in the %Na formula.
**Classification after Wilcox (1955) and Ayers & Westcot (1994)

According to the sodium percentage measurements, the irrigation water from the whole river is of "Excellent to Good" quality, with the lowest river segment displaying favourable quality at 3.6%. A %Na of 23.9% is nevertheless considered suitable for irrigation in the mid-river segment, even though it has the highest WQI because of nutrient enrichment. In order to avoid sodicity issues in the long run, FAO recommendations suggest keeping the Na concentration below 60% (Ayers & Westcot, 1994). Notably, the lower river's comparatively low %Na is in line with the WQI improvement that has been seen. According to Wetzel (2001) and Saeedi *et al*. (2010), Na levels downstream might be effectively reduced by natural attenuation mechanisms such as dilution and cation exchange activities. Sodium (Na), K, and Cl^- are soluble salts that are abundant in urban waste streams; hence, the higher levels in the mid-river may be explained by possible landfill leachate percolation (Christensen *et al*., 2001; Naveen *et al*., 2017).

While the %Na levels are still well within the acceptable range (3.6% to 23.9%) for irrigation, the adverse effects of excessive K and Pa concentrations on soil and ecosystem health in the long run are cause for worry. Despite the lack of direct sodicity concerns, prior research has shown that receiving bodies of water polluted with leachate may nevertheless experience secondary effects such as nutrient imbalances, salinity hazards, and eutrophication (Kjeldsen *et al*., 2002; Zubair *et al*., 2016). The suitability of irrigation may not be compromised by Na levels alone; however, the cumulative nutrient load highlights the need for integrated monitoring systems that take into account both the risk of sodicity and eutrophication. For agricultural activities in river water resource areas to be sustainable in the long run, mitigation methods are required. These include landfill leachate treatment and frequent soil salinity tests.

4.8.4 Composite Index (CI) assessment

The cumulative macronutrient load in the river water was evaluated according to the DWAF (1996) guideline limits using the Composite Index (CI). When the CI≤1 is of acceptable quality ie it is within permissible limits, however, CI>1 exceeds permissible limits and there is a concern for pollution. Table 4.16 displays the computed CI values, which show a clear geographical pattern. The maximum CI (1.7865) is shown in the middle of the river, which is far higher than the allowable limit. According to DWAF (1996), this indicates a notable enrichment of nutrients, mainly caused by phosphorus concentrations that are far higher than the eutrophication limit. Rivers affected by landfills are known to experience eutrophication, algal blooms, and ecological deterioration as a result of this overabundance of nutrients (Smith & Schindler, 2009; Carpenter *et al*., 1998).

Table 4.16: Composite Index (CI) for macronutrients in river water

Site	CI Value	Interpretation
Upper River	1.0463	Slightly above limits → Moderate contamination
Mid River	1.7865	Exceeds limits → Significant pollution hotspot
Lower River	0.7765	Below limits → Partial improvement downstream

The upper river showed early indications of stress and mild pollution with a CI of 1.0463, which was only slightly above the allowed level. These results are in accordance with previous research from comparable waste catchments that found that initial nutrient inputs slightly raise indices over guideline levels (Lee *et al*., 2012; Naveen *et al*., 2017). The lower river, on the other hand, had a CI of only 0.7765 which is well below the cutoff. According to Chapman (1996) and Wetzel (2001), this indicates that there may have been some dilution, sedimentation, and potential biological absorption of nutrients further downstream. Despite the positive development, the phosphorus levels are still rather high, which means that there is still a danger of eutrophication in the downstream areas overtime (Zubair *et al*., 2016).

4.9 Evaluating Possible Microbial Markers Present in the Soil and Water near the Landfill Site

4.9.1 Microbial soil analysis

Soil microbial markers are often used in environmental monitoring and agricultural activities as indicators of the biological health and fertility of the soil. All four sites of the landfill were surveyed for sample sites, with distances of 1 and 2 km measured outward from the landfill. The geographical distribution provides valuable insights into the microbiological safety over a large region, which may be used to manage food safety issues in the environment or at the perimeter. Soil quality requirements and any contamination sources may be better monitored and addressed with the use of routine microbiological testing. In order to test for microbial markers, the samples were sent to the NviroTek lab. The presence of *E. coli*, *Salmonella*, and *Listeria monocytogenes* were tested.

The results of the E. coli No Growth (NG) test indicate that the quantity of live bacteria was below the limits of detection, as shown in table 4.17 below. "No Growth" meant that the examined soil samples had a bacterial count below 10 colony-forming units (cfu/g), which is below the threshold for most soil safety regulations and is therefore safe to use. Additionally, the results show that across all soil samples there is "No Growth" (NG) for *E. coli*. Soil samples around the landfill sites are therefore not considered to be polluted with faecal material or associated pathogens if the commonly used indicator of faecal contamination, *E. coli*, is not present (Doyle & Buchanan, 2013).

Table 4.17: Soil microbial makers analysis

Location	E. coli (cfu/g)	Salmonella (per 25 g)	Listeria mono. (per 25 g)
West perimeter	*NG	Absent	Absent
North perimeter	NG	Absent	Absent
South perimeter	NG	Absent	Absent
East perimeter	NG	Absent	Absent
East 1km	NG	Absent	Absent
West 1km	NG	Absent	Absent
South 1km	NG	Absent	Absent
North 1km	NG	Absent	Absent
South 2km	NG	Absent	Absent
North 2km	NG	Absent	Absent

West 2km	NG	Absent	Absent
East 2km	NG	Absent	Absent
Control	NG	Absent	Absent

**NG indicates: No growth*

Salmonella spp. are enteric bacteria often acknowledged as markers of faecal contamination in environmental matrices. Despite being non-native to soil ecosystems, they can endure for prolonged durations in terrestrial settings when introduced by organic waste or leachate (Jechalke *et al*. 2019; Woodford *et al*. 2024). Research indicated that *Salmonella* was present in 57–88% of soil samples, varying by season, so establishing landfills as significant sources of microbial spread (Frączek *et al*. 2022). Another research showed that *Salmonella* may contaminate nearby soils via many paths (Frączek *et al*. 2022; Kalwasińska *et al*. 2013). However, the findings of this investigation show that none of the soil samples examined had *Salmonella Spp*. According to Frączek *et al*. (2022), the summer is the season when the most cases of *Salmonella* are reported, while the winter months see a decrease in cases. Leachate migration, bio-aerosol production during waste treatment, and vector animals (birds, rodents) conveying contaminated particles are potential pathways for *Salmonella* to move from landfill interiors into nearby soils (Kalwasińska *et al*. 2013). According to Frączek *et al*. (2022), the main factors that may be used as indicators are land-use and seasonality. They discovered no correlation between *Salmonella* abundance and active landfill activities, indicating that the release and persistence of enteric bacteria inside the waste material may be delayed. A study by Majowicz *et al*. (2010), the soil examined did not contain any *Salmonella*, which means it met safety criteria for this disease.

The ability of *Listeria monocytogenes* to produce severe and often fatal disease, especially among susceptible populations, makes it a major public health problem (CDCP, 2019; Stapleton *et al.,* 2024; Zahid *et al.,* 2023). The placental barrier is permeable to this virus, putting pregnant women at increased risk of complications throughout pregnancy and the newborn's health (Swaminathan & Gerner-Smidt, 2007; CDCP, 2018; Stapleton *et al.,* 2024; Zahid *et al.,* 2023). Similarly, severe listeriosis symptoms, such as septicaemia and meningoencephalitis, are more common among the elderly and those with impaired immune systems; these complications may lead to significant illness and death (ECDPC, 2023). All of the soil samples examined in this research did however not contain traces of *Listeria*, indicating that the soil is devoid of the bacterium.

Possible reservoirs for the persistence and transmission of pathogens might be created by landfill leachates and microbial migration, which could impact soil microbiota. Depending on the habitat and land use, investigations into *Listeria monocytogenes* (Lm) in both natural and agricultural soils have shown detection rates varying from around 5% to 15% (Gholipour *et al*., 2020). In older landfills, where the quantity of microbial genes, including virulence factors, rises with age, landfill leachate may transfer nutrients, organic wastes, and enteric bacteria, including Lm, into the surrounding soils (Sévellec *et al*., 2022). Abiotic and biotic variables both have a role in determining whether Lm will survive in soil (Locatelli *et al*., 2013). Endogenous soil bacteria reduce Lm survival in neutral to alkaline soils (pH>7), although Lm survival is improved in acidic or sterilised soils according to Locatelli *et al*. (2013). According to Sévellec *et al*. (2022), there is a lot of variation across strains; yet, certain Lm lineages, especially those with certain genetic markers linked to soil suitability, have survival rates over 5%. According to Chandler-Khayd *et al*. (2023), landfills have the potential to change soil physicochemical properties, which might indirectly increase the prevalence of Lm. According to studies by Locatelli *et al*. (2013), Sévellec *et al*. (2022) and Chandler-Khayd *et al.* (2023), the penetration of leachate may alter pH, nutrient levels (such as organic carbon, zinc, and calcium), and biotic competition variables. Also, according to Hui *et al*. (2023), strains in neighbouring soils may be more persistent or dangerous if older landfills, which are known to have higher levels of antibiotic resistance genes and virulence factors, are allowed to naturally select for them. Patterns of geographical dispersion are further complicated by the fact that rainfall and runoff are subject to seasonal change (Hui *et al*., 2023). In order to guarantee that ready-to-eat meals are safe, it is essential to take strong precautions against this disease (Farber & Peterkin, 1991).

These outcomes demonstrate the use of food safety management methods, such as Hazard Analysis and Critical Control Points (HACCP), that aim to reduce the likelihood of contamination (Codex Alimentarius Commission, 2009). Bacteria may still be present, but in undetectable amounts. Even though they won't grow on certain media, certain bacteria may go into a dormant condition where they might still be harmful if they were to awaken. Because soil samples were not contaminated with *Salmonella* or *Listeria* to a detectable level since these pathogens were not present in the samples. It might be owed to the soil safety management at the sample sites was successful since these key pathogens were not present. *Listeria monocytogenes*, in particular, may thrive at lower temperatures and pose serious health hazards; thus, it is crucial to have strict controls in place to avoid contamination.

4.9.2 Microbial analysis of water

The microbial examination of water is essential for evaluating water quality and establishing its appropriateness for ingestion, recreational activities, or other uses. The investigation conducted examined numerous microbiological markers throughout three segments of a river: Upper, Mid, and Lower. The bacteria analysed comprised *Listeria monocytogenes, Salmonella Typhi, Escherichia coli (E. coli),* and Total Coliforms. These microbiological markers are essential diagnostic tools for assessing environmental pollution and inferring the potential existence of pathogenic microorganisms that represent substantial threats to public health. They offer an indirect but dependable indicator of faecal contamination, facilitating the evaluation of water quality and the risk of disease transmission among exposed people.

Table 4.18 shows that none of the river segments tested positive for *Listeria monocytogenes* or *Salmonella Typhi*. Given the severity of diseases linked to these infections, this is positive results. *Listeria monocytogenes* is a food-borne disease that may cause listeriosis. People with compromised immune systems, pregnant women, and infants are most at risk (Schuchat *et al*., 1994). According to Crump *et al*. (2004), typhoid fever may be caused by *Salmonella Typhi* and can result in high fever, stomach discomfort, and may cause death if left untreated. The lack of these pathogens indicates that the river water may not be severely polluted with these particular bacteria; nevertheless, this does not exclude alternative forms of microbial contamination.

Table 4.18: Analysed microbial markers in the water from three river sections

Microbial Marker	Upper River	Mid River	Lower River
Listeria monocytogenes (Counts/25 g)	*ND	ND	ND
Salmonella Typhi (cfu/100 mL)	ND	ND	ND
Escherichia coli (cfu/100 mL)	*TNTC	TNTC	76,000
Total Coliforms (cfu/100 mL)	TNTC	TNTC	358,000

**ND (Not Detected): Indicates that the microbial marker was not found in the sample.*
**TNTC (Too Numerous to Count): Refers to counts so high that it is impractical to quantify them.*

An important sign of sewage pollution in water is the presence of *Escherichia coli* (E. coli). It is reported as TNTC in the upper and mid river sections and as 76,000 cfu/100 mL in the lower river section. The elevated E. coli concentrations in the upper and mid river indicate significant faecal contamination, maybe resulting from urban or agricultural runoff, insufficient wastewater treatment, or the direct discharge of refuse or sewage into the river. The lower river poses a significant threat to public health, still due to its high numbers (76,000 cfu/100 mL), though they are not higher than the upper and midriver sections. The World Health Organisation reports that water with E. coli bacteria concentrations more than 1000 cfu/100 mL indicates unsanitary conditions and a higher probability of water-related illnesses (WHO, 2017). The total coliform bacteria count is a measure of the water's overall quality and possible pollution. This category of bacteria is often found in the intestines of warm-blooded animals. The lower river segment had 358,000 cfu/100 mL of total bacteria, whereas the higher and middle parts of the river also recorded TNTC. It is concerning that the sampled sections of the river may not fulfil safety criteria for potable or recreational use, especially considering the high amounts of total coliforms, which raise concerns about faecal pollution.

A major public health issue is the observation of high levels of *Escherichia coli* and total coliform bacteria in the lower section of the river. It is well-established that the presence of these organisms indicates faecal contamination, which in turn indicates the potential existence of pathogenic germs that may cause infectious disorders such as gastrointestinal (Latchmore *et al*., 2023). Epidemiological studies have consistently linked E. coli and total coliforms in water sources that people consume or swim in, which may lead to a greater risk of AGI symptoms such nausea, vomiting, diarrhoea, and stomach pain (Latchmore *et al*., 2023; Strauss *et al*., 2001).

A large body of research has shown a correlation between surface water contamination from E. coli and coliform bacteria due to non-regulated agricultural runoff, poor waste disposal, and insufficient sewage infrastructure (Abdel-Shafy & Mansour 2018; Singh *et al*., 2024; Mazreku *et al*., 2025). Tetteh *et al*. (2020) investigated rivers in South Africa and discovered that coliform levels were much higher in areas where there was wastewater discharge from informal settlements and animal farms close to waterways. Similarly, Hounmanou *et al*. (2016) highlighted how untreated sewage and wastewater often end up in rivers in peri-urban regions, leading to ongoing pollution with microbes. In particular, susceptible groups, such as children and those with impaired immune systems, are at increased risk of developing gastrointestinal infections as a result of exposure to these polluted waters via consumption, skin contact, or irrigation (Ashbolt, 2004; Momba *et al*., 2006).

4.10 Chapter Summary

The comprehensive evaluation of soil, water, and microbiological indicators surrounding the landfill presents a multifaceted yet informative environmental profile. While soils generally meet international standards, they do have localised enrichments of copper, manganese, and zinc around the perimeters. The mobility and bioavailability of these metals are made easier by pH variations. On the other hand, nutrient-driven deterioration is evident in the river system. Phosphorus enrichment causes hyper-eutrophic conditions, very low Water Quality Index values, and increased composite indices that indicate ecological stress and acute eutrophication danger, all of which lead multiple indices to consistently categorise the mid-reach as a pollution hotspot. Sodium levels are still under safe irrigation standards, but the nutritional load highlights the substantial landfill runoff and leachate inputs. This contradiction is further supported by microbial tests, which show that soils are pathogen-free whereas river water shows significant levels of faeces (especially E. coli and coliform counts), which increases the hazards to public health downstream.

CHAPTER 5

SUMMARY, CONCLUSION AND RECOMMENDATIONS

5.1 Introduction

This chapter synthesises the findings of the study, which aimed to assess the environmental impacts of landfill sites on soil and water quality in Ga-Rankuwa township. The research was guided by three primary objectives: (1) to evaluate the presence of heavy metals in soils adjacent to the landfill site, (2) to assess the water quality from the river near the landfill site, and (3) to evaluate the possible microbiological markers present in the soil and water near the site. The findings are discussed in relation to the research objectives, and the implications for environmental management and public health are explored. This chapter also outlines the limitations of the study, its contributions to the field, and provides recommendations for future research and policy interventions.

5.2 Summary of Research Findings

The summaries of the findings are discussed per objective below.

5.2.1 Objective 1: Evaluating the presence of heavy metals in soils adjacent to the landfill site

The primary purpose was to evaluate the contamination levels of heavy metals in soils near the landfill site. The results showed that while the concentration of Zinc (Zn), Manganese (Mn), and Copper (Cu) differed at various sites, all of them were considered to be "uncontaminated" according to the Geo-accumulation Index (Igeo) and the permissible limits set by the World Health Organisation (WHO). However, certain sites, like the East Perimeter for Zn and the South Perimeter for Mn, had comparatively elevated values, indicating localised pollution. The presence of heavy metals in soils, even at concentrations below the contamination threshold, may nevertheless jeopardise soil health, since heavy metals adversely impact soil fertility and microbial diversity. Soil acidity increases metal mobility, according to the research, which in turn affects bioavailability and mobility of heavy metals. The solubility of heavy metals in soil near landfills is greatly affected by the pH of the soil in such places. High levels of soluble metals, which are harmful to plants and microbes, may accumulate in acidic environments like North 1 km and West 2 km. Conversely, eastern sites often exhibit higher alkaline pH values, perhaps attributable to variations in geological substrates or less exposure to landfill effluents. Soil quality was generally within acceptable levels, as confirmed by the Contamination Factor (CF) and modified degree of contamination (mCd) indices. Some sites showed moderate contamination. However, the heightened concentrations of Cu at the West Perimeter and Zn at the East Perimeter need additional examination, since these metals may accumulate over time and provide enduring concerns to environmental and human health.

5.2.2 Objective 2: Assessing the water quality from the river near the landfill site

According to the results, the landfill had a major impact on the river's water quality, especially in the middle of the river where the WQI was 422.32, which is a very low indicator of water quality. There were higher concentrations of Sodium (Na), Potassium (K), and Phosphorus (P) near the middle of the river, which are probably associated with agricultural runoff and landfill leachate. The presence of algal blooms and decreased oxygen levels in aquatic habitats may be brought about by eutrophication, which is particularly brought to light by the high levels of phosphorus (454). The river's condition is deteriorating due to nutrient runoff, according to the multi-index evaluation, which shows that the mid-reach is the most polluted area. Phosphorus-driven hyper-eutrophic conditions (TSI >100; NPI for P in the hundreds) and very poor overall water quality (mid-river WQI ≈ 422; CI ≈ 1.79) indicate high risk of algal blooms and hypoxia, while the lower reach shows partial dilution/attenuation. Although %Na values imply irrigation suitability, the scale and consistency of nutrient enrichment strongly implicate landfill-derived leachate or other anthropogenic inputs.

5.2.3 Objective 3: Evaluating the possible microbial markers present in the soil and water near the landfill site

The third objective was to detect microbial indicators in the soil and water next to the landfill site. The soil microbial examination indicated no growth (NG) of harmful bacteria, including E. coli, *Salmonella*, and *Listeria monocytogenes*, indicating that the soil samples were microbiologically safe. The lack of these diseases does not exclude the possibility of additional microbial pollutants or future contamination, particularly considering the dynamic behaviour of microbial populations in response to environmental stresses. The microbiological study of the river water indicated substantial faecal pollution, especially in the lower portion, where counts of *Escherichia coli* (E. coli) (TNTC) and total coliform (TNTC) were very elevated. Microbial testing of river water, on the other hand, showed that it was heavily polluted with human waste, especially in the upper and midriver portions where the levels of E. coli (TNTC) and total coliform (TNTC) bacteria were quite high. Although pathogens such as *Listeria monocytogenes* and *Salmonella Typhi* were not detected, the presence of high levels of E. coli and total coliforms suggests that the river water is unsuitable for consumption or recreational use without proper treatment. The disparity in microbial quality between soil and water emphasises the several entry points for contamination. Soil may serve as a barrier, stopping the spread of certain diseases, while bodies of water are more easily polluted by things like agricultural runoff and leachate.

5.3 Conclusion

This research has yielded significant insights into the environmental effects of dump sites on soil and water quality in Ga-Rankuwa township. Although the soil quality was determined to be within acceptable levels, heavy metal contamination was observed in some areas, and there was substantial water pollution, especially in the middle and lower parts of the river. According to the results, there is a noticeable difference in the quality of the soil and water close to the landfill. Soil samples obtained within a 2 km radius were devoid of E. coli, *Salmonella*, and *Listeria monocytogenes*, indicating a low microbial risk and successful contamination management in the terrestrial ecosystem. The river water samples exhibited significantly elevated amounts of E. coli and total coliforms, especially in the lower segment, where the counts reached critical levels of public health concern. Although there were no signs of *Salmonella* or *Listeria* in the water, the high levels of faeces clearly suggest that there are continuous human-caused contaminants from places like wastewater discharge, runoff, or agricultural effluents. Legislators and environmental managers can prevent secondary pollution and safeguard human and environmental health by tackling these issues and working towards long-term waste management solutions. Moreover, other heavy metals and microbial assessments not included in this study should be considered.

5.4 Recommendations

Consequently, the research presents the following recommendations:

- Systematic and routine monitoring of soil and water quality in proximity to landfill sites should be institutionalized to facilitate early detection and mitigation of contamination. Regulatory frameworks must be rigorously enforced to ensure adherence to established environmental standards, as non-compliance can exacerbate ecological degradation and public health risks with particular emphasis on limiting the leaching of heavy metals and other pollutants into surrounding ecosystems.
- Landfill operators should adopt advanced engineering solutions, such as state-of-the-art leachate collection and treatment systems, to minimise the release of harmful contaminants into adjacent soil and water bodies. Furthermore, the integration of waste segregation at source and comprehensive recycling programs should be prioritized to reduce the volume of non-biodegradable and hazardous waste entering landfills.

- Public health campaigns must be designed to educate local populations about the hazards associated with polluted soil and water. Educational activities must underscore the significance of safe water practices, including boiling or filtering water, and promote community involvement in environmental monitoring and reporting.
- Areas displaying high levels of heavy metals, especially in the East and South Perimeters as indicated in this research, must be prioritised for remediation. Established methods like phytoremediation, which uses hyperaccumulator plants to extract metals from soil, and soil washing, which utilises chemical agents to eliminate toxins, should be used to rehabilitate soil health and mitigate ecological threats.
- Long-term, longitudinal studies are encouraged and crucial for capturing seasonal and temporal fluctuations in heavy metal concentrations and microbiological activity in and around waste sites. The observed variation in soil pH and the presence of elevated E. coli and coliform levels in downstream river sections highlight the ongoing exposure risks for nearby communities. Future studies should investigate the health consequences of prolonged exposure to polluted soil and water, especially for at-risk populations living near dump sites. In this regard, interdisciplinary approaches such as epidemiological research must examine the relationship between pollutant concentrations and illness prevalence to guide public health initiatives.

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ANNEXURE A: Ethics letter 1

UNISA university of south africa

UNISA-CAES HEALTH RESEARCH ETHICS COMMITTEE

Date: 20/02/2023

Dear Mr Matlakala

NHREC Registration # : REC-170616-051
REC Reference # : 2023/CAES_HREC/002
Name : Mr IT Matlakala
Student #: 62828282

Decision: Ethics Approval from 16/02/2023 to 31/01/2026

Researcher(s): Mr IT Matlakala
62828282@mylife.unisa.ac.za; 081-409-9876

Supervisor (s): Dr MP Radingoana
radinmp@unisa.ac.za; 011-471-9760

Working title of research:

Assessing the impact of landfill sites on soil and water quality in neighbouring communities: A case study of Ga-Rankuwa township, Gauteng Province of South Africa

Qualification: MSc Geography

Thank you for the application for research ethics clearance by the Unisa-CAES Health Research Ethics Committee for the above mentioned research. Ethics approval is granted for three years, **subject to further clarification and submission of yearly progress reports. Failure to submit the progress report will lead to withdrawal of the ethics clearance until the report has been submitted.**

The researcher is cautioned to adhere to the Unisa protocols for research during Covid-19.

Due date for progress report: 31 January 2024
The progress report is available on the college ethics webpage: https://www.unisa.ac.za/sites/corporate/default/Colleges/Agriculture-&-Environmental-Sciences/Research/Research-Ethics

Please note the points below for further action:

University of South Africa
Preller Street, Muckleneuk Ridge, City of Tshwane
PO Box 392 UNISA 0003 South Africa
Telephone: +27 12 429 3111 Facsimile: +27 12 429 4150
www.unisa.ac.za

*The **low risk application** was originally **reviewed** by the UNISA-CAES Health Research Ethics Committee on 16 February 2023 in compliance with the Unisa Policy on Research Ethics and the Standard Operating Procedure on Research Ethics Risk Assessment.*

The proposed research may now commence with the provisions that:

1. The researcher(s) will ensure that the research project adheres to the values and principles expressed in the UNISA Policy on Research Ethics.
2. Any adverse circumstance arising in the undertaking of the research project that is relevant to the ethicality of the study should be communicated in writing to the Committee.
3. The researcher(s) will conduct the study according to the methods and procedures set out in the approved application.
4. Any changes that can affect the study-related risks for the research participants, particularly in terms of assurances made with regards to the protection of participants' privacy and the confidentiality of the data, should be reported to the Committee in writing, accompanied by a progress report.
5. The researcher will ensure that the research project adheres to any applicable national legislation, professional codes of conduct, institutional guidelines and scientific standards relevant to the specific field of study. Adherence to the following South African legislation is important, if applicable: Protection of Personal Information Act, no 4 of 2013; Children's act no 38 of 2005 and the National Health Act, no 61 of 2003.
6. Only de-identified research data may be used for secondary research purposes in future on condition that the research objectives are similar to those of the original research. Secondary use of identifiable human research data require additional ethics clearance.
7. No field work activities may continue after the expiry date. Submission of a completed research ethics progress report will constitute an application for renewal of Ethics Research Committee approval.

Note:
*The reference number **2023/CAES_HREC/002** should be clearly indicated on all forms of communication with the intended research participants, as well as with the Committee.*

Yours sincerely,

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Prof MA Antwi

Chair of UNISA-CAES Health REC

E-mail: antwima@unisa.ac.za

Tel: (011) 670-9391

Prof M Ntwasa

Acting Executive Dean: CAES

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ANNEXURE B: Amended ethics letter

UNISA-CAES HEALTH RESEARCH ETHICS COMMITTEE

Date: 17/04/2024

Dear Mr Matlakala

NHREC Registration # : REC-170616-051
REC Reference # : 2023/CAES_HREC/002
Name : Mr IT Matlakala
Student #: 62828282

Decision: Ethics Approval Confirmation after First Review from 16/02/2023 to 31/01/2026

Researcher(s): Mr IT Matlakala
62828282@mylife.unisa.ac.za; 081-409-9876

Supervisor (s): Dr MP Radingoana
radinmp@unisa.ac.za; 011-471-9760

Working title of research:

Assessing the impact of landfill sites on soil and water quality in neighbouring communities: A case study of Ga-Rankuwa township, Gauteng Province of South Africa

Qualification: MSc Geography

Thank you for the submission of your yearly progress report to the Unisa-CAES Health Research Ethics Committee for the above mentioned research. Ethics approval is confirmed to continue for the originally approved period, subject to submission of yearly progress reports. **Failure to submit the progress report will lead to withdrawal of the ethics clearance until the report has been submitted.**

Due date for next progress report: 31 March 2025
The progress report form can be downloaded from the college ethics webpage:
https://www.unisa.ac.za/sites/corporate/default/Colleges/Agriculture-&-Environmental-Sciences/Research/Research-Ethics

Furthermore, the following amendment is approved:

1. Collection of water samples from the nearby river system.

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Open Rubric

*The **low risk application** was originally **reviewed** by the UNISA-CAES Health Research Ethics Committee on 16 February 2023 in compliance with the Unisa Policy on Research Ethics and the Standard Operating Procedure on Research Ethics Risk Assessment.*

The proposed research may now commence with the provisions that:

1. The researcher(s) will ensure that the research project adheres to the values and principles expressed in the UNISA Policy on Research Ethics.
2. Any adverse circumstance arising in the undertaking of the research project that is relevant to the ethicality of the study should be communicated in writing to the Committee.
3. The researcher(s) will conduct the study according to the methods and procedures set out in the approved application.
4. Any changes that can affect the study-related risks for the research participants, particularly in terms of assurances made with regards to the protection of participants' privacy and the confidentiality of the data, should be reported to the Committee in writing, accompanied by a progress report.
5. The researcher will ensure that the research project adheres to any applicable national legislation, professional codes of conduct, institutional guidelines and scientific standards relevant to the specific field of study. Adherence to the following South African legislation is important, if applicable: Protection of Personal Information Act, no 4 of 2013; Children's act no 38 of 2005 and the National Health Act, no 61 of 2003.
6. Only de-identified research data may be used for secondary research purposes in future on condition that the research objectives are similar to those of the original research. Secondary use of identifiable human research data require additional ethics clearance.
7. No field work activities may continue after the expiry date. Submission of a completed research ethics progress report will constitute an application for renewal of Ethics Research Committee approval.

Note:

*The reference number **2023/CAES_HREC/002** should be clearly indicated on all forms of communication with the intended research participants, as well as with the Committee.*

Yours sincerely,

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ANNEXURE C: Data collection request letter

Request for permission to conduct research at Ga-Rankuwa landfill.

To: City of Tshwane Metropolitan

Municipality From: Dr Makgalake P.

Radingoana

Topic: Data collection request for Mr Matlakala IT (62828282)

Date: 19 July 2023

Dear Sir/Madam

I Dr Makgalake Pabalelo Radingoana, a senior lecturer at the University of South Africa hereby request permission for my student to access the Ga-Rankuwa landfill to collect data. I am an MSc supervisor for Mr Matlakala Icaboth from the University of South Africa who is conducting a study looking at the possible impacts the landfill might have on the surrounding soil and water sources. Mr Matlakala seeks to only collect water and soil samples from the area to assess how contaminated they might be so he can make recommendations on how the area can be managed and monitored to avoid continuous contamination of the environment, if any. This will also allow him to successfully complete his Master of Science in Geography.

I have worked with Mr Matlakala for over a year now on his MSc in Geography under my supervision. Throughout the year up until to date, he has proven to be a hard-working individual and has demonstrated great qualities as a postgraduate student. In addition, he has meticulously paved his way through the proposal stage, and I am more than proud to say that he managed to have his proposal approved in the first year of study.

I would further like to appreciate your sincere assistance on the above request and Mr Matlakala will also share the findings from his dissertation with the waste management department of the municipality after completion.

Kind regards,
Dr MP Radingoana
Senior lecturer/supervisor: Department of Geography
011 471 9760
Radinmp@unisa.ac.za

ANNEXURE D: Data collection approval letter

CITY OF
TSHWANE
IGNITING EXCELLENCE

City Strategy and Organizational Performance

Room RD 17 | Ground Floor, West Wing, Block D | Tshwane House | 320 Madiba Street | Pretoria | 0002
PO Box 440 | Pretoria | 0001
Tel: 012 358 4209
Email: IsaiahE@tshwane.gov.za | www.tshwane.gov.za | www.facebook.com/CityOf Tshwane

My ref:	**Research Permission/ Matlakala**	Tel:	012 358 4559
Contact person:	**Pearl Maponya**	Email:	PearlMap3@tshwane.gov.za
Section/Unit:	**Knowledge Management**	Date	24 July 2023

Mr Tshwarelo Matlakala
1921 Block X Extension
Mabopane Unit X EXT
0190

Dear Mr Matlakala,

RE: ASSESSING THE IMPACT OF LANDFILL SITES ON SOIL AND WATER QUALITY IN NEIGHBOURING COMMUNITIES: A CASE STUDY OF GA-RANKUWA TOWNSHIP, GAUTENG PROVINCE OF SOUTH AFRICA

Permission is hereby granted to Mr Tshwarelo Matlakala, Master of Science in Geography degree candidate at the University of South Africa (UNISA) , to conduct research in the City of Tshwane Metropolitan Municipality.

It is noted that the aim of the study is to assess the environmental impacts of landfill on soil and water quality in Ga-Rankuwa township. The City of Tshwane approves the use of its name in the study and further notes that all ethical aspects of the research will be covered within the provisions of UNISA Research Ethics Policy. You will be required to sign a confidentiality agreement with the City of Tshwane prior to conducting research.

Relevant information required for the purpose of the research project will be made available as per applicable laws and regulations. The City of Tshwane is not liable to cover the costs of the research. Upon completion of the research study, it would be appreciated that the findings in the form of a report and or presentation be shared with the City of Tshwane.

Yours faithfully,

PEARL MAPONYA (Ms.)
DIRECTOR: KNOWLEDGE MANAGEMENT

City Strategy and Organisational Performance • Stadstrategie en Organisatoriese Prestasie • Lefapha la Thulaganyo ya Tiro le Togamaano ya Toropokgolo • UmNyango wezokuSebenza namaQhinga aHleliweko kaMasipala • Kgoro ya Leanopeakanyo la Toropokgolo le Bodiragatši bja Mmasepala • Muhasho wa Vhupulani ha Ḓorobo khulwane na Mashumele • Ndzawulo ya Maqhinga ya Dorobakulu na Matirhele ya Masipala • Umnyango Wezeqhinga Ledolobha Nokusebenza Kwesikhungo