EFFECTS OF WASTE DUMPING ON WATER QUALITY, SOIL AND PLANT DIVERSITY AROUND CYUVE DUMP SITE IN MUSANZE CITY, RWANDA

BY

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DECLARATION

This dissertation is my original work and has not been presented for a degree in any other University

………………………………… ……………………………

Signature Date

APPROVAL

We as University supervisors confirm the work done by the candidate under our supervision.

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Signature Date

DEDICATION

I dedicate this dissertation to my beloved family, including Odette NIYONAGIZE, for their unwavering support and encouragement throughout my academic journey. I also extend my heartfelt gratitude to my teachers and mentors for their invaluable guidance and wisdom, which have been instrumental in my success.

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LIST OF ACRONYMS

ANOVA: Analysis of Variance

Cd: Cadmium

CEC: Cation Exchange Capacity

CH_4: Methane gas

Cm: Centimeter

CO_2: Carbon Dioxide

COD: Chemical Oxygen Demand

Cr: Chromium

DO: Dissolved Oxygen

EC: Electrical Conductivity

H_2: Hydrogen

H_2O: Water

HMs: Heavy Metals

LSD: Least Significant Difference

m: Meter

mL: milliliter

MSW: Municipal Solid Waste

MSWM: Municipal Solid Waste Management

NH_3: Ammonia

OM: Organic Matter

Pb: Lead

RSB: Rwanda Standard Board

TDS: Total Dissolved Solids

TOC: Total Organic Carbon

TVC: Total Viable Count

WHO: World Health Organization

ABSTRACT

The rapid urban population growth and increasing demand for resources have led to a global surge in waste generation, a trend also evident in Musanze City. At the Cyuve dumpsite, waste accumulation poses potential threats to soil quality, surface water, and plant biodiversity. This study aimed to evaluate the effects of the Cyuve waste dumpsite on the surrounding ecosystem, particularly focusing on soil and water quality and plant diversity. A quantitative research design was employed, with soil samples collected from three distances downstream (0–40 m, 40–80 m, and 80–120 m) and at three depths (0–5 cm, 5–15 cm, and 15–30 cm), whereas the control site was located at 100 meters upstream the dumpsite. Surface water samples were taken from three stream locations nearby to the dumpsite. Samples were analyzed during both wet and dry seasons, and statistical analysis was performed using RStudio version 4.3.1. Results from ANOA and LSD test ($p \leq 0.05$), showed that soil closer to the dumpsite (0–40 m) had higher mean values for pH, organic matter (OM), electrical conductivity (EC), total dissolved solids (TDS), and cation exchange capacity (CEC), which decreased with distance. The study revealed that heavy metal concentrations, including lead, chromium, and cadmium, were highest in the soil samples collected closest to the Cyuve dumpsite. Lead levels ranged from 0.19 to 0.88 mg/kg during the wet season, with slightly elevated concentrations of 0.37 to 1.49 mg/kg in the dry season. Similarly, chromium levels varied between 2.62 and 5.44 mg/kg in the wet season, increasing to 3.38 to 6.13 mg/kg in the dry season. Cadmium concentrations followed the same pattern, ranging from 0.353 to 0.54 mg/kg in the wet season and rising slightly to 0.193 to 0.79 mg/kg during the dry season. Surface water analysis revealed increased bacterial contamination (Total Viable Count, Total Coliforms, *Escherichia coli*), and heavy metal concentrations downstream and middle stream, exceeding safe limits set by the Rwanda Standards Board and WHO. Despite these findings, plant diversity remained high across all distances, indicating a heterogeneous plant community. The study concluded that the Cyuve dumpsite is negatively influencing soil and water quality, posing potential health risks. The study highlights the urgent need for Musanze City to promote public awareness and education on waste reduction, recycling, and reusing materials. Community engagement is essential for fostering responsible waste management. Additionally, proper waste management solutions, such as engineered landfill sites, are crucial to prevent environmental degradation and protect natural resources.

CHAPTER ONE

INTRODUCTION

1.1 Background of the study

Waste is broadly defined as any discarded material generated from various sources such as waste management facilities, air contamination regulator facilities, manufacturing processes, commercial activities, mining operations, and agronomic practices (Salam, 2017). This encompasses a wide range of substances including trash, rubbish, slurry, and other rejected materials, whether they are in solid, fluid, semisolid, or gaseous form. Generators of waste are typically classified into different categories as outlined by Khan *et al*. (2016), including domestic, industrial, commercial, offices, construction, public, and agronomic sources. Domestic waste, often referred to as municipal waste, stems from households and communities. Industrial waste, on the other hand, arises from factories and industrial processes, often containing harmful substances. Additionally, medical waste, considered hazardous due to its infectious nature, falls under a specialized category of waste materials.

Currently, there is a significant rise in waste worldwide, primarily ascribed to various reasons, including rapid population expansion rising living standards, and increased urbanization (Mehmet *et al*, 2022). It was outlined in the study conducted by Song *et al*. (2014), that a rise in waste production is exacerbated by the expanding global population, a growing economy, accelerated urbanization, and the elevation of living standards in communities worldwide. The quantity of waste generated globally is incredible, with an estimated annual production reaching around 11 billion tons, and the per capita waste production averaging approximately 1.74 tons per year (Song et al., 2015). This large volume of waste reflects a concerning trend of resource degradation driven by the increasing demand for new products.

A study conducted by Mavakala *et al.* (2022) noted that municipal solid waste production in developing countries is expected to double or even triple by the year 2050, constituting a substantial 35% of global waste. Oteng-Ababio *et al.* (2013) highlighted the significant challenges faced by communities and leaders, particularly in Sub-Saharan African towns, in managing waste effectively. One major contributing factor is the lack of awareness and adherence to proper waste disposal practices within society. Furthermore, the insufficient political will and commitment from leaders exacerbate the waste management problem (Oteng-ababio et al., 2013). This lack of responsiveness hinders the implementation of efficient waste management strategies. Consequently, many developing countries resort to disposing of solid waste in open dumpsites, primarily due to the availability of cheap land. However, this practice has detrimental consequences for ecosystem dynamics in the long term.

Waste dumping is defined as the act of releasing, adding, placing, dropping, or retaining discarded materials or hazardous waste from community activities on a specific area of land or into a body of water (Salam, 2010). A considerable amount of the waste produced globally is dumped in engineered landfill sites or open dumpsites. It was noted by Bikash *et al*. (2015) that open dumpsites are the primary method used in developing countries for waste disposal due to their economic feasibility and simplicity. Approximately 95% of the waste collected globally is either dumped in engineered landfills or open dumpsites (Bikash *et al*., 2015). While alternative solid waste management methods such as waste reutilization, composting, and incineration are available, approximately 10 to 20% of the waste generated from these alternatives still requires disposal in engineered landfills or open dumpsites (Tamru & Chakma, 2016).

In many low-income countries, waste disposal remains a significant challenge, often leading to environmental contamination due to inadequate measures to address toxic gas emissions and leachate production (Bundhoo, 2018). Siddiqua *et al.* (2022) noted that a dumpsite refers to an open area where waste is simply disposed of, whereas a landfill is a properly engineered structure built either into or on the ground. A sanitary landfill also known as engineered landfill, is designed to safely contain waste, isolating it from groundwater and maintaining dry conditions. However, in most developing countries, insufficient infrastructure, limited financial resources, lack of technical expertise, and low societal awareness contribute to inadequate waste management practices (Mavakala *et al.*, 2022). Consequently, open dumpsites become the primary method of waste disposal, posing significant environmental and health risks by producing leachate.

Leachate is generated in waste dumpsites when moisture, such as rainwater or snow, infiltrates the waste, percolating through and dissolving various substances. The decomposition of organic materials by microbial activity and chemical reactions within the waste, such as oxidation, further contribute to leachate formation. As waste piles compact over time, liquids are squeezed out, increasing the overall volume of leachate (Zhang et al., 2020).

The infiltration of water from open dumpsites promote the migration and distribution of dissolved and suspended solids in the soil, eventually contaminating groundwater or adjacent surface water bodies. This process alters the physicochemical properties of these water sources. Therefore, the uncontrolled dumping of waste in open dumpsites can have profound impacts on various components of the ecosystem, including soil, water, plants, living organisms, and human health (Mavakala *et al.*, 2022). The significant effects of environmental contamination on human health are particularly evident in developing nations, where an estimated 90% of deaths are attributed to environmental pollution (Khan *et al.*, 2016). Ecosystem issues such as land degradation, water pollution, and air contamination are closely linked to inadequate waste management practices.

In East Africa, countries face significant challenges in managing this growing waste burden effectively. The rapid urbanization and population growth in cities such as Kampala, Nairobi, and Dar es Salaam have resulted in an overwhelming surge in waste generation (Nabegu, 2010). These cities often lack the necessary infrastructure, financial resources, and efficient waste management systems, leading to improper waste disposal practices. As a result, much of the waste in these areas is disposed of in open dumpsites, which creates serious environmental and public health risks, particularly through the production of leachate that can contaminate groundwater and surface water bodies (Henry *et al.*, 2006).

Rwanda faces significant challenges in establishing effective waste management systems, from the point of generation to final disposal or processing. The current methods remain inadequate and unsustainable, failing to adhere to best practices for waste management. In the Cities of Rwanda including Kigali, the solid waste management system relies on open dumping at landfills, which is becoming a growing concern for municipalities across the country. These dumping sites are major contributors to pollution, affecting air, water, and land quality (Iraguha *et al.*, 2022).

Open dumpsites remain a frequently used way for disposing of waste in various places of East African countries, including Rwanda, primarily because of their low cost and economic feasibility. However, this practice has severe environmental consequences, as water infiltration through the waste promotes the spread of harmful substances, contaminating both groundwater and surface water bodies (Bundhoo, 2018). The long-term impacts of such improper waste disposal are evident in the degradation of soil, water, plant life, and overall ecosystem health (Mavakala *et al*., 2022).

The Cyuve dumpsite in Rwanda is a prime example of how improper waste disposal can lead to environmental contamination. Studies in assessing the environmental impacts of waste dumping are critical and proposing effective remediation strategies. The main goal of the present study is to investigate the effects of waste dumping at the Cyuve dumpsite, with a particular focus on its impact on the physicochemical properties of soil, water quality, and plant species diversity. Given the detrimental effects of heavy metals and other contaminants, primarily originating from waste, on environmental quality, it has become essential to regularly conduct assessments to propose effective remediation approaches (Agbeshie *et al*., 2020). Thus, the present study aims to investigate the effects of waste dumping on the physicochemical properties of soil, water quality, and plant diversity specifically at the Cyuve dumping site.

1.2 Problem statement

Management of waste has become a critical global challenge, particularly in urban areas where rapid population growth and industrialization have resulted in a surge in waste generation. Municipalities in developing countries, responsible for managing municipal solid waste (MSW), struggle to provide efficient and adaptive waste management systems. These challenges are due to inadequate collection methods, improper disposal practices, limited landfill capacity, a lack of technical expertise, and insufficient financial support (Srivastava et al., 2015). Inadequate waste management is a leading contributor to environmental pollution, negatively impacting air quality, soil health, and water resources, thereby posing a risk to both ecosystems and human health (World Bank, 2018).

In East Africa, waste management challenges are no less significant. The region faces an increasing volume of solid waste due to urbanization and economic growth, yet it lacks the necessary infrastructure to manage this waste effectively. Many cities across East Africa, including Kampala, Nairobi, and Dar es Salaam, encounter similar obstacles such as insufficient waste collection services, the absence of engineered landfills, and the lack of proper waste treatment facilities.

According to a study by Okot-Okumu (2021), open dumping remains the predominant method of waste disposal in these cities, contributing to widespread environmental degradation and public health risks. The situation is exacerbated by the limited availability of financial and technical resources needed for efficient waste management.

Like many other cities across the world, Musanze, a secondary city in Rwanda, encounters considerable obstacles in terms of solid waste management. These challenges encompass insufficient waste collection methods, a lack of appropriate waste disposal tools, inadequate waste disposal procedures, and overloaded disposal sites. These issues are made worse by the enormous amount of waste produced every day in Musanze, which severely affects the city's infrastructure and resources. A study by Ngwijabagabo *et al*, (2020) noted that Musanze City produces about 150 tons of waste daily, but only 12 tons are disposed of at the Cyuve public dumpsite, which has now exceeded its capacity. Located near a residential area, the dumpsite sits just 5 meters above the water table, as confirmed by hydrogeological assessments (Ngwijabagabo *et al*., 2020). The Cyuve waste dump site receives waste from a variety of sources, including household, commercial, and industrial activities, making it a primary dumping ground for the Musanze city's waste.

The Cyuve waste dump site represents a significant environmental concern as it operates as a non-engineered open dump lacking essential features such as a bottom liner or a leachate collection and treatment system. This is aggravated by the region's notable precipitation levels, with an average annual rainfall ranging between 1184-1800 mm per year, as documented by Twahirwa et al. in 2023. Considering these weather conditions, it is highly likely that leachate, along with its contaminants, seeps from the dump site into the nearby soil and water streams.

This migration of leachate from the dump sites has potential effects on the physicochemical properties of the receiving soil and water. In addition to that, it contains high amounts of hazardous compounds, including substances known to be carcinogenic, posing risks to the environment and human beings (Wuave, 2021). Alteration in the physicochemical composition of water and soil may adversely affect plant diversity, pH balance, and the availability of nutrients (Zhao et al., 2022). Thus, the migration of leachate in soil has the potential to disrupt the balance of ecosystems, leading to adverse ecological impacts. Yet, there is a lack of adequate data concerning the level of soil and water contamination, along with the status of plant diversity near the Cyuve waste dump, thereby requiring additional investigation. Hence, the main objective of this study was to assess the influence of the Cyuve waste dump site on the physicochemical characteristics of soil and water, as well as the diversity of plant species in the vicinity.

1.3. Objectives

1.3.1. General objective

The general goal of this investigation was to evaluate the effects of the leachate from Cyuve waste dumpsite on the adjacent surface water quality, soil and plant diversity.

1.3.2. Specific objectives

1. To assess the effects of leachate on the physical and chemical characteristics of the soil downstream from the Cuyve dump site, and the levels of heavy metals within it.
2. To determine the effects of leachate on the physical and chemical properties, concentrations of heavy metals, and bacterial levels in the stream water near the Cyuve dump site.
3. To evaluate the diversity of plant communities downstream of the Cyuve dump site.

1.4. Research questions

1.4.1. What are the effects of the leachate from Cyuve waste dumpsite on the physicochemical properties and concentrations of heavy metals in the nearby soil?

1.4.2. What are the effects of the leachate from Cyuve waste dumpsite on the physicochemical properties, concentrations of heavy metals, and bacterial loads of the nearby water stream?

1.4.3. What is the effects of Cyuve dumpsite on the diversity of the nearby plant community?

1.5. Significance of the study

This study aimed to assess the concentration levels of heavy metals in soil and establish the soil's physicochemical properties. Additionally, it seeks to provide data on plant diversity and surface water quality parameters around the Cyuve dumping site. Furthermore, the scientific insights gained from this study will assist policymakers and conservationists in developing informed, effective, and sustainable measures to address environmental pollution. This aligns with the United Nations' Sustainable Development Goal 3, which aims to promote good health and well-being for all. Moreover, soil remediation actions based on the scientific findings will support the achievement of the fifteenth Sustainable Development Goal, encouragement life on land.

1.6. Scope of the study

The aim of this research was to examine the physical and chemical characteristics of soil downstream from the Cyuve dump site, as well as the levels of heavy metal contamination within it. Furthermore, the study sought to analyze the water quality of the dump site nearby stream, and to investigate the diversity of plant species downstream from the Cyuve dump. The aim of this research was to examine the physical and chemical characteristics of soil downstream from the Cyuve dump site, as well as the levels of heavy metal contamination within it. Additionally, the study sought to analyze the water quality of the nearby stream influenced by the dump site, and to investigate the diversity of plant species downstream from the Cyuve dump. The research was conducted over a period of 12 months, allowing for comprehensive seasonal data collection and analysis.

1.7. Conceptual Framework

The movement of leachate from the Cyuve municipal solid waste dumpsite may influence nearby soil physical and chemical characteristics, as well as the levels of heavy metals, surface water quality, and the diversity of plant communities.

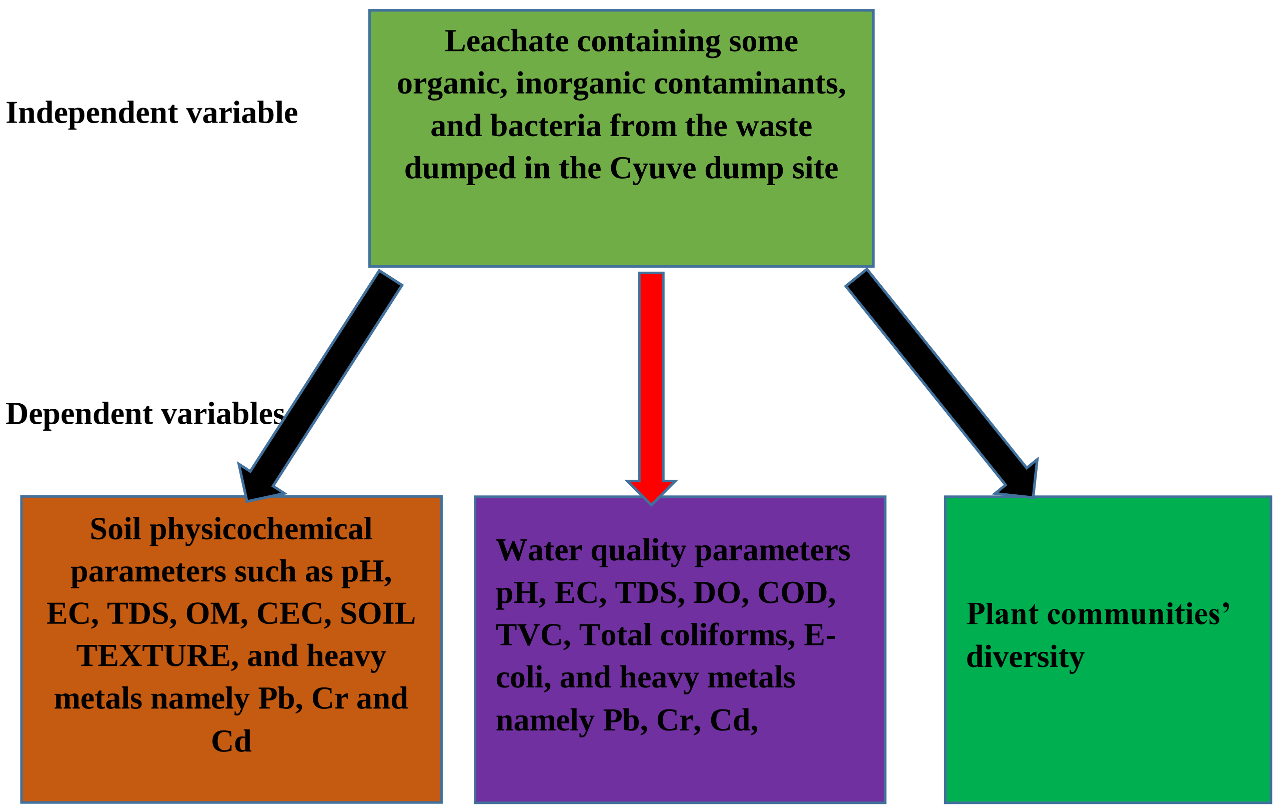

Figure 1.1: Conceptual Framework of the study

Through phytoextraction, plants absorb contaminants, such as heavy metals, from the soil through their roots and storing them in their tissues. Over time, this accumulation can affect plant health, reduce species diversity, and alter the ecosystem by favoring species that are more tolerant over others, leading to decreased overall diversity near contaminated sites like the Cyuve dumpsite.

CHAPTER TWO

LITERATURE REVIEW

2.1 Introduction

This section thoroughly examines the published research work that have been done on waste and waste management, with an emphasis on the processes that take place in landfills (dump sites), soil, and water systems. The focus is on clarifying the complex ways in which these processes affect ecosystem dynamics. This chapter includes a special section that looks at how leachate affects soil composition and subsequently plant diversity.

2.2. Literature on solid wastes disposal

2.2.1 Waste generation

Plant leaves, leftover food, unwanted papers, clothes, ash, dust, stones, animal carcasses, human and animal excreta, waste from construction sites, glass, plastic containers, metal scraps, plastic bags, condoms, and expired agricultural pesticides and insecticides are examples of solid wastes commonly deposited in landfills. In addition, other types of waste such as construction and fabrication remnants, hazardous waste, sludge, radioactive materials, food processing byproducts, and household hardware including furniture and electrical materials contribute to landfill accumulation.

The qualities of a community's solid waste are influenced by various factors such as working conditions, living standards, lifestyle, and income levels. A study conducted on solid waste characterization in East African Community cities revealed that a significant portion of the waste is organic, with data indicating a substantial difference in the percentage of organic waste between low and high-income communities. This highlights how changes in living standards greatly influences the composition of solid waste generated. The analysis further underscores the strong correlation between income levels and waste composition, particularly in regions with a predominance of low to low-middle income populations, such as the East African Community (Gilbert *et al.*, 2021). A similar study by Hoornweg and Bhada-Tata (2012) also found that lower-income communities tend to produce more organic waste, due to a reliance on agriculture and food markets, whereas higher-income communities generate more plastics and other non-biodegradable waste. As a result, a larger proportion of organic waste is produced in lower-income areas, signifying the potential for easier composting due to the decomposable nature of organic waste. The table 2.1 below illustrates the proportions of waste generated globally across communities with different income levels:

Table 2.1: Proportions of wastes generated from different communities

Income level	**organics %**	**papers %**	**Plastics %**	**Glasses %**	**Metals %**	**Others %**
Low income	64	5	8	3	3	17
lower middle income	59	9	12	3	2	15
upper middle income	54	14	11	5	3	13
High income	28	31	11	7	6	17

Source: (Gilbert et al., 2021)

As Table 2.2 shows, organic waste makes up a high percentage of solid waste in East African nations. While some of these organic wastes decompose readily and can be composted, others decompose more slowly and require disposal in landfills. In addition, a significant portion of the waste generated in these countries comprises materials that can be incinerated, recycled, or otherwise managed. The degree of industrialization in East African nations can be linked to the occurrence of organic waste, as stated by Gilbert *et al.* (2021). The food processing industry is quite low in East African countries, in contrast to developed nations where it greatly decreases organic waste. Consequently, a considerable portion of food is consumed fresh, leading to the production of significant amounts of organic waste.

Table 2.2. Composition of solid waste produced in major cities across East Africa

Type of waste (%)	**Dar es salaam/ Tanzania**	**Kampala/ Uganda**	**Kigali/ Rwanda**	**Nairobi/Kenya**
Organic waste	71	77.2	68	65
Paper	9	8.3	9	6
plastic	9	9.5	5	12
Glass	4	1.3	-	2
Metal	3	0.3	2	1
Others	4	3.4	15	14

Source: (Gilbert *et al*., 2021)

According to Mavakala *et al*. (2022), the primary characteristic of the developing nations is inefficient solid waste management, which may be attributed to inadequate financial resources, technical skills, community attitudes, and proper infrastructure and planning. This was in agreement with the study conducted by Oteng-ababio *et al*. (2013), who noted that waste collection is especially deficient in impoverished urban communities, where people deal with piles of waste going uncollected for weeks until to the time of burning it or disposing of it in stagnant gutters and streams, all of which attract pests that spread disease and pose a serious risk to the public's and the environment's health.

This is consistent with the findings reported in the study by Gilbert *et al*. (2021), who noted that a relatively little percentage of East African Community solid wastes are recycled and that many of them are landfilled or discarded in an unsustainable manner as well as thrown in inappropriate locations. According to the study conducted by Oteng-ababio *et al*. (2013), the issue is frequently exacerbated by the residents of these communities' ignorance and lack of complacency with regard to appropriate disposal practices. Many techniques have been developed recently to minimize the quantity of waste produced, with an emphasis on reducing the amount of waste that is not needed. Some goods, including paper, plastics, metals, aluminum, and glass, are reprocessed. Reusing certain old tools is another strategy that has been implemented to reduce the requirement for raw materials. Moreover, waste can be converted into electricity, which will help reduce the amount of disposed waste (Tursunov *et al*., 2020).

Iravanian and Ravari (2020) noted some particular strategies that have been devised to lessen the production of waste. Some methods include reducing non-recoverable waste, recycling materials such as glass, steel, metal, aluminum, plastics, and paper, and reusing materials to decrease the need of raw material and energy consumption. However, certain substances remain undesired and useless, and their disposal must be done in a way that reduces deterioration of the environment. Therefore, disposing of solid waste must be done safely, and this choice should only be made in the absence of other options (Vaverková, 2019)

2.2.2 Waste disposal

In the earlier part of the 20th century, waste disposal practices often neglected considerations for environmental contamination. Dumpsites, primarily in the form of open landfills, were utilized with the assumption that groundwater and soil would naturally manage any leachate produced by the waste (Meegoda *et al*., 2016). However, this approach proved to be problematic as these sites lacked essential mechanisms such as liners to collect leachate, resulting in its percolation into the surrounding soil (Abelson, 1984). Despite concerns raised by European nations as early as the 1930s regarding the standardization of waste disposal methods, little action was taken until 1959 (Vallero & Blight, 2019). During this period, it was commonly believed that the ecosystem could handle the impact of waste disposal without significant short-term or long-term consequences. This mindset led to the increase of poorly designed landfill sites, exacerbating the issue of environmental contamination.

In 1959, the American Society of Civil Engineers introduced the concept of engineered landfills as a response to growing concerns about waste management. Later, in 1979, the US Environmental Protection Agency (EPA) implemented regulations targeting the reduction of negative environmental and public health effects linked to waste disposal sites (Iravanian and Ravari, 2020). These regulations marked a significant step towards addressing the shortcomings of waste disposal practices. According to these guidelines, a landfill would be categorized as an open dump if it did not meet the specified criteria set forth by regulatory bodies. Recognizing the need for improved waste management, the United States' Resource Conservation and Recovery Act (RCRA) has advocated for the closure or modification of open dumpsites to align with the requirements of engineered landfills. Moreover, there has been a concerted effort to discourage the continued use of open dumpsites in favor of more environmentally sound waste disposal methods (Smith et al., 2015).

The majority of countries have put regulations in place to convert open dumpsites to the engineered landfills, but others continue to dispose of waste in an outdated manner that allows insects, greenhouse gasses, rats, and leachate from those open dumpsites to contaminate ecosystems (Amoah & Kosoe, 2014). Therefore, in the engineered landfills, wastes are compacted in a designed landfill using a large, steel-wheeled machine, and the area is then covered with a layer of soil every day after work (Meegoda *et al*., 2016). Based on the research carried out by Abelson (1984), a sanitary landfill must have certain features, such as leachate collection liners, impermeable clay, and covering, to prevent pollutants from infiltrating into soil. Spreading, compacting waste, lining systems, and daily soil covering processes are all used in these engineered sanitary landfills.

2.3. Landfills classification

Although there are other landfill characteristics that can be utilized to categorize landfills, Resource Conservation and Recovery Act mentions the most widely used approach (RCRA, 2014). Every category of landfill is made to accommodate a certain kind of waste and operates accordingly. Landfills are categorized by the type of waste they receive, as per RCRA. As mentioned in the study conducted by Iravanian and Ravari (2020), solid wastes in this classification system do not always have to be in the form of physical solids; they can also be in liquid, semisolid, or gaseous state.

Based on the research carried out by Meegoda *et al*. (2016), Solid wastes are commonly classified into two main categories: hazardous and non-hazardous. Non-hazardous solid wastes constitute the first major group, suitable for disposal in various types of landfills. Among these, the first category comprises landfills designated for municipal solid wastes. Furthermore, a study conducted by Meegoda et al. (2016) showed that, under certain circumstances, household waste as well as other non-hazardous materials like sludge, industrial solid waste, and construction and demolition wastes (C&DW) can be dumped in municipal solid waste landfills. The second category of non-hazardous landfills receive the industrial waste. Various industrial processes, including electricity production, fertilizer or agricultural chemical manufacturing, food processing, inorganic chemical production, iron and steel manufacturing, leather production, nonferrous metal manufacturing or foundries, organic chemical production, plastics and resins manufacturing, paper production, stone, glass, clay, and concrete product manufacturing, textile production, transportation equipment manufacturing, and water treatment, produce industrial wastes that need to be managed and appropriately dumped (Iravanian & Ravari, 2020).

The second major group of landfill is that designed to receive hazardous unwanted materials. Resource Conservation and Recovery Act of the united states has defined hazardous waste as waste that can have destructive impacts on the human body and ecosystems. Hazardous wastes come from many different sources such as industrial practices that produce or recycle batteries, fluorescent light bulbs, cathode ray tubes, diverse medicinal and radiological waste (RCRA, 2014). Generally, this type of waste comprises some pollutants that are more toxic and heavy metals with long half-lives like barium, lead, cadmium, and mercury (Iravanian & Ravari, 2020). Thus, the constituent of deposited waste is an important parameter in landfill categorization.

2.4. Phases of waste decomposition

When solid wastes are deposited in dumpsites, a series of processes occur to facilitate their decomposition. A research carried out by Ratna *et al*. (2021) noted that solid waste deposited in landfills absorbs water during rainfall. As the waste compresses under the weight of the percolated water, a liquid called leachate is generated, comprising a diverse range of organic and inorganic compounds. According to the study conducted by Iravanian and Ravari (2020), waste stabilization occurs through five distinct stages, which are as follows:

1. **The initial stage of adjustment**

During this stage, the landfill becomes moist as waste are buried. It takes time for adequate moisture to enhance microbial activities. To create an atmosphere that is conducive to biochemical decomposition, some preliminary changes to the environment are made.

2. **Transition stage**

During this stage, the landfill's aerobic environment changes to an anaerobic one. At the end of this stage, noticeable amounts of volatile organic acids (VOAs) and chemical oxygen demand (COD) are identified in the leachate components.

3. **The stage of acid generation**

During this stage, large amounts of intermediate volatile organic acids are generated due to the continued hydrolysis of waste and the microbial decomposition of biodegradable matter. pH value declines at this phase

4. **Stage of methane fermentation**

Methane-forming consortia consume intermediate acids during this stage, converting them to carbon dioxide and methane. Then cause the growth of methanogenic bacteria due to the increase of pH.

5. **Stage of Maturation**

Biological activity and gas production are decreased during this stage, less nutrients and substrates are available. Due to low concentrations, the leachate strength remains constant at this stage.

2.5. Leachate composition

The amount and quality of leachate in dumpsites are determined by numerous parameters namely the quantity of waste, components in the waste, how those components are soluble in water, and the amount of humidity in waste. Omofunmi *et al.* (2020) observed that leachate properties are affected by a range of factors, including solid waste composition, landfill operational techniques, climatic and hydrogeological factors, as well as internal landfill conditions such as biochemical processes, moisture content, temperature, pH, and landfill age. Samadder *et al.* (2017) urged that the constituents of leachate differ from one dumpsite to another depending on waste components, decomposition stage, and dumping system. This was supported by Mehmet and Acarer, (2022) who stated that the components of leachate differ due to category, age, humidity, the degree of degradation of the waste in that dumpsite, height, the dumping method, and weather conditions of the landfill play a vital role in leachate composition. It was noted in the study conducted by Vodyanitskii (2016) that the components of the leachate produced from the waste in the dump sites are subdivided into four groups namely dissolved organic matter, inorganic constituents, heavy metals, and xenobiotic organics.

2.5.1. Dissolved organic matter

One of the components found in the leachate from landfill is dissolved organic matter (DOM). Based on the study carried out by Kjeldsen *et al.* (2002), the dissolved organic matter (DOM) includes volatile fatty acids and other more resistant components including humic-like and fulvic-like compounds, and other components such as total organic carbon (TOC), chemical oxygen demand (COD), and Biological Oxygen Demand (BOD). Scandelai *et al.* (2019) stated that many different organic compounds like alcohols, phenols, aldehydes, and ketones have been discovered in leachates collected from many areas of the world. In addition, Huo *et al*. (2008) indicated that both organic and inorganic pollutants can interact with dissolved organic matter (DOM) where this interaction of DOM and other elements in the environment occurs due to certain functional groups in DOM, such as carboxylic, phenolic, and carbonyl. Furthermore, according to a study conducted by Vodyanitskii (2016), some of those organic contaminants are very mobile, difficult to break down, and do not get adsorbed by soil particles. As a result, they can disrupt aquatic life, pH of the water, and lower the water’s quality when they enter water sources.

2.5.2. Inorganic Components

Several ions, including Ca^{2+}, Mg^{2+}, Na^{+}, K^{+}, NH^{4+}), Fe^{2+}, Mn^{2+}, Cl^{-}, SO4 $^{2-}$ and HCO^{3-}, NO_3^{2-} and others, are contained in the inorganic component of the leachate (Kjeldsen et al., 2002; Vodyanitskii, 2016). The levels of inorganic macro components in the leachate is dependent on the stages of landfill stabilization, according to Iravanian and Ravari (2020). In methanogenic reactions, the pH tends to be elevated, while the concentrations of calcium, magnesium, iron, and manganese decrease. On the other hand, the level of sulfate also decreases because of microbial reduction of sulfate to sulfide. A research carried out by Kjeldsen *et al.* (2002) noted that ammonia is the leachate's most important element. Proteins breaking down contribute to ammonia production in landfills. Yet, there's currently no system to decrease ammonia levels, which only decrease if released from the landfill via leachate. Research indicates that ammonia concentration remains consistent even three decades post-closure of the landfill (Kjeldsen et al., 2002).

2.5.3. Heavy metals

Cadmium (Cd^{2+}), chromium (Cr^{3+}), copper (Cu^{2+}), lead (Pb^{2+}), nickel (Ni^{2+}) and zinc (Zn^{2+}) are some of the heavy metals that are present in the leachate (Kjeldsen *et al.*, 2002; Vodyanitskii, 2016). Leachate contaminated with heavy metals is a significant concern among various contaminants. Batteries, consumer electronics, ceramics, light bulbs, glass, and food industry wastes are among the materials that Iravanian and Ravari (2020) revealed to be dumped in landfills. Numerous substances contain a diverse range of heavy metals in relatively small amounts, some of which pose significant hazards and can be extremely harmful even at low concentrations. Additionally, Naveen *et al.* (2018) reported that since organic acids tend to enhance the dissolution of heavy metals in low pH, heavy metal concentrations are generally higher in the early stages of waste decomposition. As per the research conducted by Naveen *et al*. (2018), heavy metals can contaminate soil, groundwater, and surface water. Their ability to move, dissolve, and transfer in water or plants can also adversely affect human health.

Table 2.3: Maximum permissible limits for heavy metals in uncontaminated soil

Heavy metals mg/kg	EU STD mg/kg	UK STD mg/kg	US STD mg/kg	WHO mg/kg	Ranges for uncontaminated soil mg/kg (Nangia, 2001)
Cadmium	3	1.4	400	0.002-0.5	0.01 - 0.7
Chromium	180	6.4	400	0.002-0.2	5 - 3000
Lead	300	70	300	0.3 - 10	2 - 200

EU = Europe, UK= United Kingdom, US = United States, WHO = World Health Organization, STD = Standard.

Source: (Ediene & Umoetok, 2017)

2.5.4. Xenobiotic organic compounds

As described in the study conducted by Kjeldsen *et al.* (2002), xenobiotic organic compounds include halogenated hydrocarbons like tetrachloroethylene and trichloroethylene as well as monoaromatic hydrocarbons like benzene, toluene, ethylbenzene, and xylenes. According to a study conducted by Vodyanitskii (2016), these compounds are typically found in industrial and domestic chemical wastes, including those containing pharmaceuticals, industrial, pesticides, and pharmaceuticals. Furthermore, an additional origin of xenobiotic organic compounds found in landfills includes substances like food additives such as stabilizers, antioxidants, pigments, and materials used in food packaging. This aligns with the findings of the conducted study by Samadder *et al.* (2017) who reported that xenobiotic leachate constituents normally come from domestic and manufacturing wastes like cosmetic oil, medications, factories, and agronomic wastes.

Xenobiotic organic compounds, according to Iravanian and Ravari (2020), have complex chemical structures that persist in the environment for a long period of time, and can affect the physicochemical properties of agricultural soils, groundwater, surface water, and aquatic life in a short amount of time. Thus, the long-term environmental consequences of these compounds may include toxicity and biological accumulation in the cells of living things. On the other hand, Kjeldsen *et al.* (2002) indicated that leachate originating from waste landfills may contain additional components such as borate, sulfide, arsenate, selenate, barium, lithium, mercury, and cobalt that are typically present in relatively low quantities

2.6. Effects of leachate on soil

As outlined in the study conducted by Zhou (2017), the bulk density, texture, structure, and pore space of soil are all determined by the proportions of minerals, air, water, and organic matter in the soil. Chemical reactions in soil also take place on soil colloids due to their big surface area and charges. Colloid surfaces have the ability to capture, hold, and release ions according to their size, charge, and concentration. Since organic acids can dissociate to create a net negative charge in the soil, it has a cation exchange capacity that is four to fifty times higher than clay per unit mass in soil (Zhou, 2017). This suggests that organic matter is a source of negative charges.

Generally, precipitation significantly influences the generation of leachate and migration. The rainwater percolates into wastes dumped in the landfill sites and increase the decomposition process, and then convey the contaminants into the soil through infiltration. This has been supported by Sam-uroupa and ogbeibu (2020) who said that leachates might get into the soil from dumpsites by surface runoff, leaching, and percolation, to develop the harmful effects on environmental compartments by altering the physicochemical characteristics of the soil and water over time.

Therefore, the release of H^+ ions resulting from the breakdown of organic materials contributes to increased soil acidity. Furthermore, creation of carbonic acid via the reaction between carbon dioxide (CO2), generated from decaying organic matter, and soil moisture further reduces the pH of the soil. Yet, a conducted study by Domínguez *et al.* (2019) on acidic soils amended with solid wastes observed the opposite effect. According to their findings, the use of composts derived from urban waste on soil commonly resulted in an increase in pH. This occurrence was linked to the moderating impact of organic matter and the existence of carbonates within the waste materials. Leachate predominantly contains elevated levels of dissolved organic and inorganic ions (S. Mehmet & Acarer, 2022). These different chemical substances if they are not well treated will subsequently affect the electrical conductivity of the surrounding soil by increasing the salts concentrations and ions from wastes to the soil. It was indicated by Nta and Odiong (2017) that the electrical conductivity value below 0.5milliScm-^{1} is harmless to soil without any negative impact on the plant development.

Natural soil usually contains some dissolved organic matter, according to Iravanian and Ravari (2020). The amount varies depending on vegetation, soil composition, clay mineral content, metal oxide presence, and environmental conditions such as precipitation and temperature. However, allowing leachate to seep into the soil raises its dissolved organic matter and puts the soil ecology out of balance. Furthermore, the existence of specific inorganic ions, like sodium, can notably diminish soil permeability. Moreover, biochemical complexes in organic substances that are glycine, citric acid, tartaric acid, and gluconic acids contain chelating properties which help them to bond with heavy metals and form tight chelates with humic acid that increases the adsorption to inhibit the mobility of heavy metals in soil. However, when heavy metal is associated with organic acids such as acetic acid facilitates the movement and obtainability of that heavy metal in soil

(Evangelou *et al*., 2007). Therefore, the evident impact of organic matter on heavy metal in soil is to accelerate or delay heavy metal mobility based on metal-organic matter formed compound.

Annabi *et al*. (2007) noted that both immature and mature solid waste composts can improve soil aggregate stability. However, this has been criticized by Harun *et al*. (2013) who indicated that the interactions between waste and soil influence the soil's structure, liquid limit, plastic limit, compaction characteristics, and strength properties, and result in many different geotechnical challenges such as landslides, ground subsidence, etc; and then affect underground structural stability. Hargreaves *et al*. (2008) also noted that waste composts contain high salt levels that can affect soil structure and disturb plant metabolism.

2.7. Effects of leachate on water resources

The leaching of inorganic and organic contaminants from open dumpsites can impact the structural qualities, chemical composition, and microbial activity of the nearby water resources. A study conducted by Kola-Olusanya (2012) noted that contamination happens when rainfall seeps into the landfill, dissolving the soluble portion of the waste and the solute fraction created by the molecular interactions and biological processes involved in the decomposition of waste. In addition, Parvin and Tareq (2021) noted that during the rainy season, surface water bodies and surrounding lowlands are contaminated by water that has leachate from the landfill site.

A study carried by Iravanian and Ravari (2020) indicated that the transfer of dissolved organic matter (DOM) from soil to water sources may alter the pH, photochemistry, and biological activity of the water as well as the concentration of oxygen dissolved in water due to the decomposers that use oxygen to decompose those dissolved organic matter. Thus, the amount of dissolved oxygen decreases when more DOM enters into water sources. This is backed by research carried out by

Kusari (2019) that showed the significant influence of the Prishtina dump site on the physicochemical characteristics of the nearby river called Sitnica, including the increase of ammonia and nitrates, phosphates, suspended material, electrical conductivity, and biological oxygen demand (BOD5). In addition, a study conducted by Vodyanitskii (2016) reported there was a decrease of microbial community structure with the decline of leachate concentration and was dependent on the distance from the point of leachate entry in the water stream.

Runoffs from municipal dump sites, uncontrolled defecation, and improper disposal of waste have all been linked to a notable rise in total and faecal coliforms in the dump sites nearby surface waters, as noted in a study carried out by Nartey *et al*. (2012). A study by Parvin and Tareq (2021) noted that heavy metals, among other constituents of landfill leachate, are non-biodegradable and can degrade the quality of groundwater and surface water, and are hazardous to biological systems even in low concentrations. In addition to being poisonous, bio-accumulative, persistent, and endocrine disruptive, heavy metals can cause cancer (Parvin & Tareq, 2021). As per the study carried out by Chu *et al*. (2019), the presence of heavy metal concentrations in leachate poses a significant threat to water contamination, influencing aquatic life, the food chain, and public health.

2.8. Effects of leachate on plant diversity

Municipal solid wastes contain many inorganic and organic impurities that can be absorbed by plants Ali *et al*. (2013). These contaminants in solid waste can also change the chemical composition of soil and pose some effects on the soil microorganisms and plants that depend on the soil for their survival (Mokobia *et al.*, 2006). As plant diversity depends on soil for nutrient absorption, they accumulate heavy metal ions through the root system. According to Nawab *et al*. (2022), contaminants may gather and transmit from soil to plant, air to crops, and water to crops.

Therefore, the soil-plant interface might contribute to the significant buildup of metals in plants. This corresponded with the study conducted by Ali *et al.* (2013) who noted that solid waste contaminants disturb the soil quality properties and eventually reduce its productivity, whereby those contaminants get into plants through root absorption and start affecting the usual plant functioning that is unapparent damage and continue till the noticeable damages get observed on some plant organs.

A study by Nawab *et al*. (2022) indicated variations in toxic element concentrations across different parts of plants, with higher bioaccumulation and absorption rates noticed in plant roots in contrast to other plant tissues. A study carried out by Chibuike and Obiora (2014) noted that the growth of cultivated crops in polluted soil could be significantly impacted by the level of heavy metal concentrations. This was in conformity with Khan *et al.'*s (2008) study, which identified elevated metal concentrations in various plant parts, including both vegetative and non-vegetative tissues, grown in soil irrigated with wastewater. López-Millán *et al*. (2009) also observed that prolonged exposure to metal concentrations primarily affected the roots and shoots of lettuce plants, potentially hindering biomass and growth by inhibiting the photosynthetic process.

The accumulation of heavy metals within plants can lead to physiological changes which in turn may inhibit plant growth and decrease yield (Khan *et al.,* 2008). A research conducted by Khan *et al*. (2008) indicated that high levels of heavy metal toxicity in plants can lead to inhibition of seed germination, significant growth rate decreases, changes in photosynthetic efficiency, respiration, and transpiration, as well as alterations in the uptake rates of essential nutrients like manganese (Mn), potassium (K), magnesium (Mg), and calcium (Ca). Toxic elements may interfere with plant growth, cell division, and other developmental processes. Moreover, heavy metals displace many essential nutrients at cation exchange sites, rendering them unavailable to plants, which has indirect negative effects (Nawab *et al.*, 2022).

Plants may also be affected by the gases from dumpsites. Kjeldsen *et al.* (2002) noted that the surrounding soil may become contaminated by the polluted gasses from the dumpsites, replacing the oxygen in the soil and affecting soil microbes and plant roots. Furthermore, heavy metals have the ability to interfere with soil microbes' ability to function, which could negatively affect plant development. This is because there are less beneficial soil microbes present, which reduces soil fertility and inhibits the composition of organic matter.

Iravanian and Ravari (2020) reported that elevated levels of various inorganic ions, including calcium, magnesium, sodium, potassium, ammonium, iron, chloride, sulfate, nitrate, and bicarbonate in the soil can cause osmotic pressure, which in turn may delay plant growth and harm vegetation. Additionally, excessive concentrations of some particular ions may injure plants and contaminate groundwater and end up in food chain. Furthermore, the heavy metals from the landfills may also find their way into the food chain and pose a risk to consumers. This aligns with the study by Amano *et al*. (2021), which highlighted various symptoms and illnesses linked to the accumulation of metals in the body, such as neurological disorders, asthma, depression, gastrointestinal issues, and cardiovascular diseases, among others. These conditions are associated with exposure to leachate components exceeding permissible limits.

2.9. Hazards arising from mismanagement of the dumpsites

Poorly managed dumpsites pose significant environmental and health hazards by contaminating groundwater, degrading soil, and releasing toxic gases like methane and carbon dioxide. These hazards, exacerbated by inadequate waste treatment, are linked to the tragic collapse of the Kiteezi dumpsite in Kampala, Uganda, which polluted local water sources and caused mortalities. Such incidents highlight the broader environmental risks associated with improper landfill management (Kigongo, 2024). In addition to physical dangers, mismanaged dumpsites like Kiteezi also generate harmful emissions, particularly methane, which can lead to explosions. Some reports suggest that a methane gas explosion may have played a role in the Kiteezi landslide, further endangering nearby residents. These incidents disproportionately affect vulnerable populations living near the dumpsite. The tragedy points to an urgent need for stricter waste management regulations and sustainable practices, such as recycling and composting, to mitigate such hazards and prevent further disasters.

2.10 Summary of literature and research gaps

Several studies have brought to light to the worrying practice of cities disposing of waste in open dumpsites without proper consideration for their proximity to residential neighborhoods, roadsides, drainage networks, riverbanks, and woodland areas. This neglect can eventually lead to the introduction of various chemicals, including hazardous elements, into different ecosystems. However, there is a need for scientific investigations on levels of heavy metals and the status of physicochemical properties in the soil, along with assessments of water quality near waste disposal facilities. A research carried out by Mekonnen *et al*. (2020) also suggested for further researches should be carried out to assess the levels of heavy metals present in both soil and water sources surrounding the waste disposal sites.

In addition, a study that was conducted on the utilization of Geographical information system (GIS) in solid waste management, in Musanze city recommended the revision of waste management procedures to take into account the city's expansion and population growth, and replacement of the Cyuve dump site due to its failure to fulfill the criteria for an appropriate landfill site (Mihigo, 2021). However, this study did not provide scientific evidence on how Cyuve dumpsite affected the surrounding ecosystem, particularly soil, surface water, and plant diversity.

CHAPTER THREE

MATERIALS AND METHODS

3.1 Introduction

This section comprises the study area description, research and sampling design, sample collection instruments, and data collection procedures that were adopted in assessing the impact of open waste disposal on the quality of soil, surface water, and vegetation diversity.

3.2 Description of the study area

The Cyuve open dump site, which is situated in the *Rwanda's Northern Province, Musanze district, Rusizi Sector, Bukinanyana Cell, and Bubandu village,* is the study's area. The site's location is approximately 68 kilometers away from Kigali. The waste piles cover an estimated 4.6 hectares and are situated at an altitude of 1844 meters above sea level, at a latitude of - 1° 30' 23.4324" S and a longitude of 29° 36' 37.6128" E. One of Rwanda's five chosen secondary cities is Musanze City, which has a dense population of 476,522 people living in 530.3 km^2 (or 898.6 people per km^2), according to the national institute of statistics of Rwanda (NISR, 2022). Additionally, this city is experiencing rapid development primarily due to the growth of tourism-related activities (Ngwijabagabo et al., 2020b). Musanze district has a humid climate with two peak rainfall seasons occurring in April and November, separated by dry spells. Rainfall averages between 1184mm to 1800mm annually. The temperatures are relatively cool, with daytime warmth and cooler nights. Average minimum temperatures drop to 13°C, reaching as low as 10°C in higher altitudes. The average maximum temperature is around 20°C (Twahirwa *et al.*, 2023). The predominant soil types in Musanze District are mainly lateritic, humus-rich, and volcanic (Rugazura & Murugesan, 2015).

Musanze City, one of Rwanda's largest and fastest-growing cities, is a major trading center, business, and tourism, which contributes to the high production of waste (Rugazura & Murugesan, 2015). Ruhondo Lake, the Buhanga eco-park, and the famous mountain gorillas in Musanze make the city a well-liked travel destination for both domestic and foreign tourists. It was noted in research carried out by Ngwijabagabo *et al*. (2020) that Musanze City produces approximately 150 tons of waste daily, with only 12 tons being disposed of at the public dumping site in the Cyuve area. Presently, the dumpsite has surpassed its maximum capacity for waste accommodation. Situated close to a residential area, the Cyuve dump site is above the water table depth of 5 meters, as indicated by various hydrogeological assessments (Ngwijabagabo *et al*., 2020a). The factors mentioned above are the reasons to conduct this study of assessing the effects of disposed waste on the nearby soil, surface water, and plant diversity at the Cyuve dumping site.

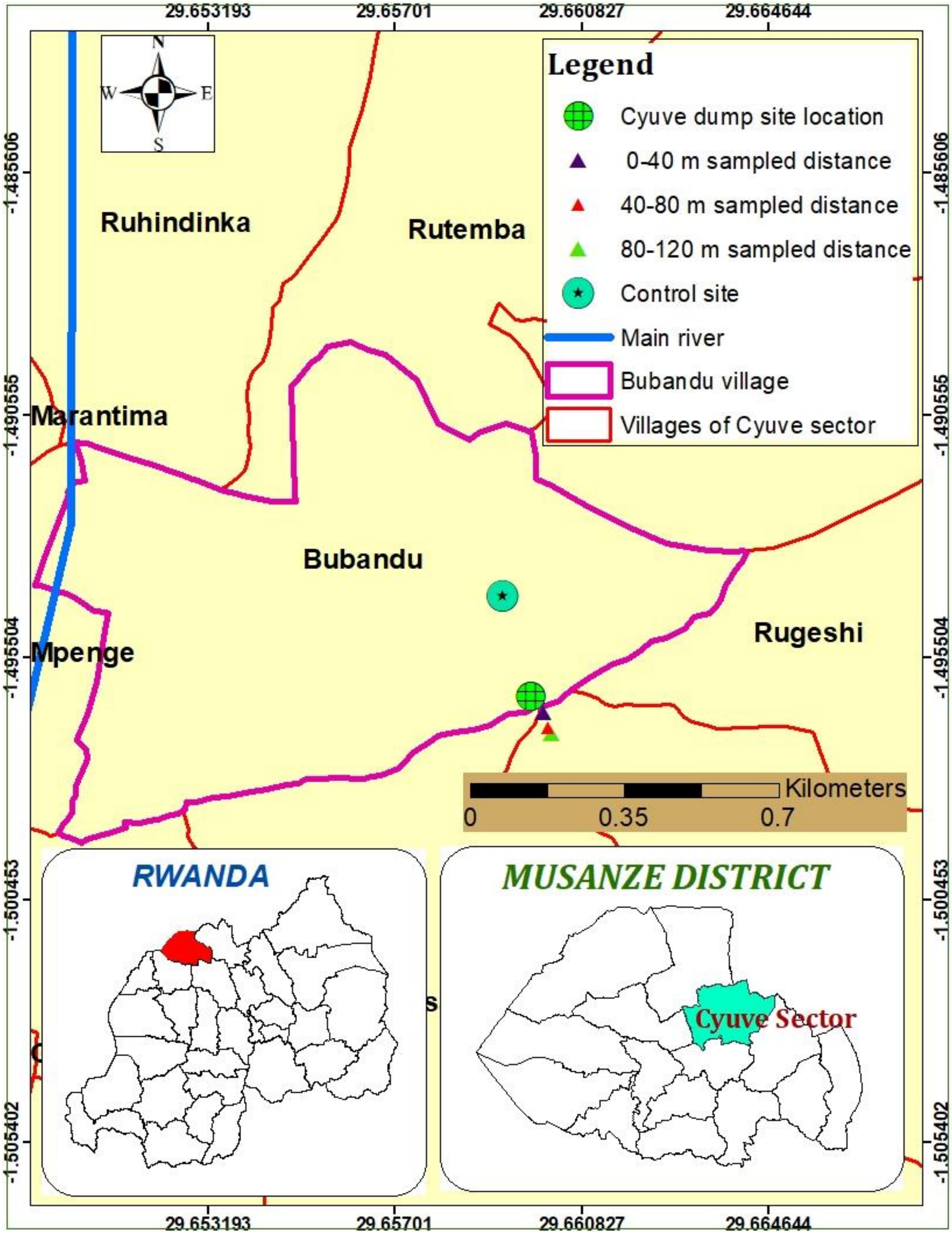

Figure 3.1: Location map of Cyuve dump site in Musanze city

3.3 Research design

The current study utilized a quantitative research methodology. It's important to highlight that quantitative research aims to measure and assess variables in order to obtain findings (Apuke, 2017). This approach can offer ways to explain a problem by gathering data in numerical form. It entails the utilization and examination of numerical data through particular statistical methods.

3.4 Data collection tools and methods

3.4.1 Soil sampling

The objective of this research was to evaluate the possible effects of leachate on soil, water, and plant biodiversity downstream the Cyuve open dumpsite in both the rainy and dry seasons. In May 2023, during the wet season, samples of leachate, soil, water, and plant were collected downstream of the Cyuve waste dump site, followed by collection in August 2023 for the dry season. The sampling frame of the study was based on stratified and simple random sampling approach, while water and leachate samples were gathered employing purposive sampling techniques, and vegetation survey with a systematically located quadrat method for vegetation data collection. The purpose of randomization in soil sampling was to avoid bias.

Based on soil sampling guidelines, the present study employed spatially explicit management which is a technic of dividing the field into smaller sections to be treated individually to increase precision (Ackerson, 2018). Soil samples were obtained at intervals of 40 meters from the edge of the dumpsite, covering distances from 0 to 40 meters, 40 to 80 meters, and 80 to 120 meters downstream. Each segment spanned the width of the dumpsite, which was 80 meters. Therefore, the soil samples were collected within an area measuring 40 meters by 80 meters. These distances were considered as factors representing the variation in horizontal distance. The sampling distances chosen at the Cyuve dumpsite (0-40m, 40-80m, and 80-120m) are essential for understanding how pollutants, particularly heavy metals and leachates, migrate horizontally through the soil profile. Shorter distances help capture contamination gradients more accurately near the source of pollution, as soil contamination typically decreases with distance due to processes like dispersion and natural attenuation. Studies by El-Fadel et al. (2002) and Agamuthu (2013) emphasize the importance of short intervals around dumpsites for closely monitoring contamination levels, which might not be evident over longer distances. These distances are crucial for understanding how pollutants, particularly heavy metals and leachates, migrate horizontally through the soil profile near the dumpsite.

The study also accounted for vertical variation by collecting soil samples from different depths using a soil auger, which was driven into the soil to obtain subsamples from varying depths (0-5 cm, 5-15 cm, 15-30 cm). Selected soil depths (0-5 cm, 5-15 cm, and 15-30 cm) were chosen because pollutants, especially heavy metals, tend to accumulate in the upper layers of soil. These layers are directly exposed to atmospheric deposition and surface runoff from dumpsites. Research by Lu et al. (2011) and Mwamburi (2013) shows that the top 30 cm of soil typically holds the highest concentrations of contaminants. Shallow depths are preferred to assess contamination risks and track pollutant infiltration before it reaches deeper groundwater layers. The simple random method was employed to collect fifteen independent sub- samples of soil at each depth (0 – 5 cm, 5 -15 cm, 15 – 30 cm) from each designated distance. Therefore, fifteen subsamples collected from the same depth range (0 – 5 cm, 5 -15 cm, 15 – 30 cm) were placed in their own bucket and thoroughly mixed. After achieving homogeneity, the bulk of the subsamples were reduced using a quartering technique to obtain one kilogram (1kg) of composite sample from each depth (Chon, n.d.).

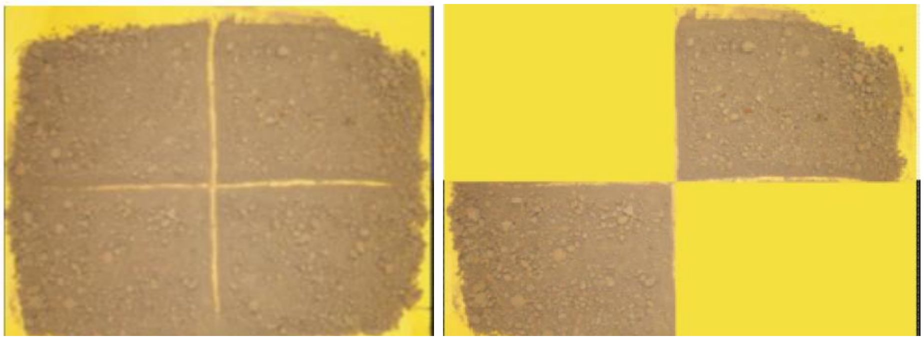

Figure 3.2: Illustration of quartering technic to reduce the bulk of soil subsamples to composite sample

The selection of fifteen (15) sampling points at each specified distance interval (0 – 40m, 40 – 80 m, 80 – 120 m) was guided by soil sampling protocols, which recommend that each composite sample should comprise 10 to 20 evenly distributed subsamples collected from various locations across the field (Ackerson, 2018). According to these guidelines, one composite soil sample should be collected for an area less than 20 acres. Based on that, the present study considered a distance of 40 meters in length and 80 meters in width, which is the width of the Cyuve dumpsite, creating an area of 0.32 hectares, which falls below the recommended maximum area outlined in the soil sampling guidelines. This approach facilitated a more precise measurement of contamination levels within a small area, compared to taking a single composite sample from a larger area.

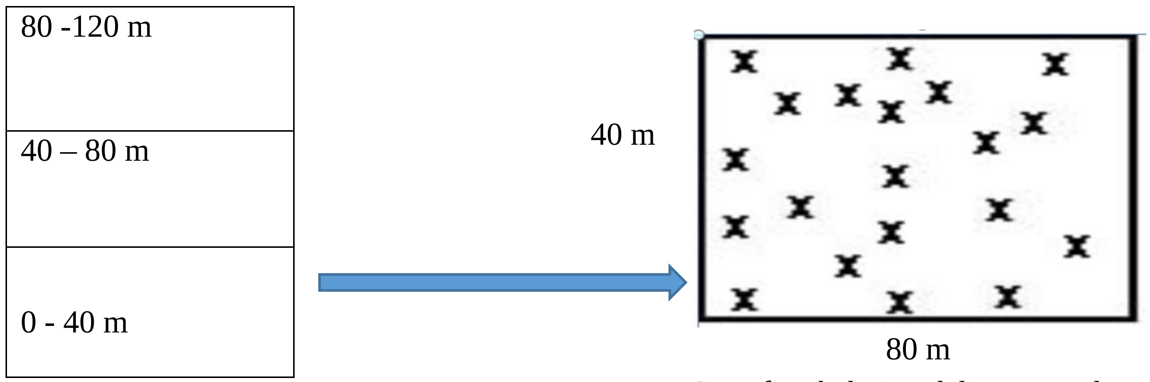

Figure 3.3: Illustration of the designed distances starting at the edge of dumping site

Size of each designed distance and sampling points.

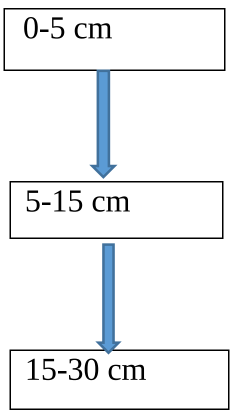

Figure 3.4: Illustration showing the depths tested within each distances

The one-kilogram composite sample obtained at each depth within the designated distances was collected in a suitable polythene bag and labeled accordingly. Thus, three (3) composite samples were gathered from three varying depths: one from 0 to 5 centimeters, the second from 5 to 15 centimeters, and the third from 15 to 30 centimeters. This process was repeated at each depth to obtain second composite samples (replicates) for biological replication, ensuring that statistical analysis of soil parameters would be based on the mean values of the replicates. To create a comparison of soil physicochemical properties among the investigated distances, and control site, present study implemented the identical soil sampling procedures at a chosen reference site (Control site). This control site was positioned at 100 meters upstream of the dumpsite, where it was expected to remain unaffected by leachate from the dumpsite, given that leachate typically drains downstream of the dump site.

Locating the control site was based on the recommendations by the U.S. Environmental Protection Agency to collect soil control samples, from a location not affected by the pollutants of concern. This agency also recommended that a control sample must be obtained as close to the sampled area as possible and from a similar soil type (U.S. EPA, 2020). Starting at 100 meters upstream the dump site, the present study designed a distance of 40 meters and considered the length (80 m) of the dumpsite as the width to make an area of 40m x 80 m for the control site. Following the same simple random sampling method employed at the investigated distances around the dump site, three (3) composite soil samples and their replicates were obtained at three different depths (0 – 5 cm, 5 – 15 cm, 15 – 30 cm) from the control site.

Therefore, the study collected a total of 24 soil composite samples for each season. These included first set of nine (9) composite samples collected from different depths within the investigated distances downstream the dump site, and three (3) from the depths of the control site, along with their respective replicates for biological replication purposes. So, this means among these twenty-four (24) composite samples collected in wet season, six (6) were collected at the control site. All soil composite samples collected were carefully packaged in appropriate polyethylene bags, labeled, and transported to the analytical laboratory of the Rwandan Agricultural and Animal Resources Development Board (RAB) for analysis.

3.4.2 Sample preparation

Soil samples were allowed to dry naturally in the air at room temperature (about 25°C) for ten days, slightly grounded and sifted using sieves with openings of 2 mm and 0.5 mm. For the analyses of soil texture, pH, electrical conductivity (EC), total dissolved solids (TDS), cation exchange capacity (CEC), and heavy metals such as lead (Pb), Chromium (Cr), and cadmium (Cd), the fraction of less than 2 mm was used. Whilst organic matter (OM) was analyzed using the percentage of less than 0.5mm. The laboratory materials, apparatus, and control conditions were used with high precision, caution, and accuracy to avoid errors and biases.

3.4.3 Measurement of soil physicochemical parameters and heavy metals

The soil samples underwent analysis to determine their physical and chemical characteristics namely soil pH, OM, EC, TDS, CEC, Soil texture, Cd, Cr and Pb, using the following standard analytical methods:

3.4.3.1 Determination of pH and Electrical Conductivity

Ten grams (10g) of dry soil sieved at 2 mm was mixed with twenty-five milliliters of distilled water using a mechanical shaker for one hour. Subsequently, the solution was left to settle for thirty minutes. At this point, pH of the soil and its electrical conductivity were determined utilizing a portable WTW pH 3110 meter and an OHAUS conductivity meter model ST 3100C, respectively.

3.4.3.2 Determination of soil organic matter

The soil organic carbon was measured using the titration method and Walkey-black processes as described by Sleutel *et al.* (2007), which was then transformed into organic matter. Using a solution of potassium dichromate and sulfuric acid, organic matter was combusted at a temperature of around 1250 degrees Celsius in this technique. The remaining or unused K2Cr2O7 is titrated using ferrous ammonium sulfate. The quantity of $K_2Cr_2O_7$ that was utilized, calculated as the difference between the initially added amount and the residual quantity, provided an indication of the organic carbon content in the soil. This organic carbon value was then converted into organic matter by multiplying it by 1.724.

3.4.3.3 Textural analysis of soil samples

In order to identify the soil texture classes of the sampled distances and control site, the BOYOUCOUS densimetric method as described by Orhan and Kılınç (2020) was utilized. A quantity of fifty (50) grams of air-dried soil that had been sieved to a size of 2 mm was put to a conical flask along with 100 milliliters of 10% hydrogen peroxide. Afterward, the mixture was subjected to heat in a sandy bath for three hours in order to completely destroy any organic matter and stop the generation of gazes. After cooling, the mixture was poured into a 1000 ml cylinder, to which 50 ml of sodium hexametaphosphate 10% was added, and the remaining space was filled with potable water.

The cylinder was placed on a level surface, then the suspension had been mixed by ten reversals, and the thermometer and hydrometer were immediately inserted into the cylinder to record the initial reading that correlated with the concentration of silts and clays. In order to get second readings that only showed the concentration of clay, the suspension was left to stand for three hours in order to insert again the hydrometer and thermometer. The percentages of sand, silt, and clay were calculated to indicate the textural class of the soil in investigated distances (0 – 40m, 40 – 80 m, 80 -120 m) and control site using the following formulas:

$$\text{Clays + Silts} = \frac{(\text{First lecture} - 2) * 100}{50\ \text{gr}}$$

Sand%=100 – (Clays + Silts) %

$$\text{Clay } \% = \frac{(Second\ lecture+2)*100}{50\ gr}$$

Silt % = 100 – sand % - clay %

Whereby first lecture is the hydrometer reading obtained directly after ten successive reversals.

Second lecture is the second reading on the hydrometer after three (3) hours.

The constant 2 must be deducted from the first lecture while calculating the percentage of (clays + silts) when the temperature recorded by the inserted thermometer is greater than 24 degrees Celsius, and it must be added to the second lecture in the calculation of clay percentage (%).

3.4.3.4 Determination of Total dissolved solids

Ten grams (10g) of dry soil sieved at 2 mm was mixed with twenty-five milliliters of distilled water using a mechanical shaker for one hour. The solution was then permitted to settle for a duration of thirty minutes. The electrode of a ST 3100C OHAUS TDS meter was inserted into the settled solution to measure the total dissolved solids in soil.

3.4.3.5 Determination of Cation Exchange Capacity (CEC)

Soil CEC was analyzed in the laboratory as described by Kitsopoulos (1999). Five grams of dried soil, sifted through a 2mm sieve, were weighed and placed onto a funnel with filter paper positioned on a 100ml volumetric flask and percolating 100ml of 1N NH_4-acetate pH 7 in portions of 10 ml ten times keeping the interval of ten minutes. The percolated extract of 1N NH_4-acetate pH 7 from the sample was collected into the flask and poured and wash the sample with 100ml of ethanol 95% in ten (10) portions of ten (10) ml to remove excess ammonia.

The residue on the filter paper was then leached using 100 ml of 1N KCl, in ten leachings of 10 ml each, with ten minutes' intervals between two successive leachings. From the extract of 1N KCl collected into a volumetric flask, the study pipetted 25 ml into a distillation tube. Afterwards, 10 ml of 40% NaOH was added, and the mixture was distilled using a Kjeldahl distiller machine until 100 ml of distillate was obtained in the receiver. The receiver contained 5 ml of boric acid with four drops of indicator. The received distillate was titrated with standard acid 0.1N HCl to the color change. Cation Exchange Capacity was then calculated using the formula provided below:

$$\text{CEC(meq/100g of soil)} = \frac{(\mathrm{V} - \mathrm{B}) * \mathrm{NHCl} * 100 * 100}{wt * 25}$$

Whereby

V: Volume of titrant used

B: Blank

N HCl: Normality of HCl acid

100: Volume of HCl leached from the sample and collected into a volumetric flask

25: Volume of pipetted extract from the 100ml collected after leaching

W: weight of sample in grams

100: 100g of soil to put the answer in mill equivalent per 100g

3.4.3.6 Determination of heavy metal concentrations in soil samples

To quantify the concentrations of heavy metals in soil, the study utilized methods outlined by Minkina *et al.* (2018). Five grams (5g) of each dry soil sample, previously sieved to a size of 2 mm, were weighed and placed on a funnel containing filter paper, positioned above a 100 ml volumetric flask. Subsequently, 100 ml of 1N NH4-acetate pH 7 (1N CH3COONH4) was percolated through the soil samples in ten sections of 10 ml each, with a fifteen-minute interval between each percolation. The percolated extract of 1N NH4-acetate pH 7 collected into labeled volumetric flasks was then taken to the Atomic Absorption Spectrophotometer (AAS) to quantify the concentrations of lead (Pb), chromium (Cr), and cadmium (Cd) in the soil samples. Using the atomic absorption spectrophotometer in an acetylene flame, the concentrations of heavy metals in the obtained extracts were measured. The instrument (AAS) was calibrated using standard solutions to measure Pb, Cr, and Cd at wavelengths of 283.306 nm, 357.869 nm, and 228.802 nm, respectively.

The choice of Pb, Cr, and Cd is particularly relevant due to their high toxicity and potential health risks. Lead exposure is linked to neurological issues, chromium to respiratory problems, and cadmium to kidney damage (ATSDR, 2012). These metals are also subject to stringent regulatory standards, emphasizing the need for monitoring in contaminated sites (EPA, 2020). Their historical association with industrial activities and waste disposal further underscores their significance as indicators of environmental health, making them critical in assessing the contamination levels downstream of the Cyuve dumpsite (Zhang et al., 2016). Using Atomic Absorption Spectrophotometry (AAS) to quantify lead (Pb), chromium (Cr), and cadmium (Cd) in soil samples is highly advantageous due to its sensitivity, specificity, and rapid analysis capabilities, making it suitable for detecting trace levels of heavy metals, which can have significant environmental and health impacts (Skoog et al., 2018). AAS measures specific wavelengths for each metal, minimizing interference from other elements and ensuring accurate quantification (Miller, 2014).

3.4.4 Water and leachate sampling procedures

Water samples were obtained from a stream water nearby to the dumpsite. Three different stations were identified for sampling purposes. Firstly, the upper stream (US), positioned at 100 meters before the stream crosses the dumping site, was nominated as the control station. The middle stream (MS) served as the second station and was situated very close to the dumping site, after the entry point of leachate into the stream water. Lastly, the downstream (DS) station was located 100 meters after the point where leachate entered the stream.

The water sample collection procedure followed guidelines for grabbing samples from rivers, streams, and freshwater wetlands (Danielson, 2014). At each station, the sampler positioned on stones at the edge of the water stream where there was a laminar flow of water to ensure sample uniformity. Additionally, the sampler held an uncapped plastic bottle of 0.5 liters upside down and submerged it in the water stream to prevent the entrance of floating debris. Afterwards, the bottle was tipped upright before stirring up the stream bottom, allowing it to be filled with water before being taken out and covered with its cap. It is important to note that for sampling water intended for heavy metal analysis, the 0.5-liter bottle was tipped upright when it was nearly stirring up the stream bottom, as heavy metals tend to accumulate in deeper waters.

Furthermore, this procedure was repeated on each station in order to collect the second samples (replicates) either for elemental analysis or heavy metal analysis for biological replication purpose. This indicates that from each station, the study sampled two bottles of water samples for elemental analysis and two bottles of water samples for heavy metals analysis in each season. Two milliliters of concentrated HNO3 acid were added to the bottles containing water samples for heavy metal analysis to bring the pH down to less than two, preventing the adsorption of heavy metals on the bottom of the sample containers as described by Mekonnen *et al.* (2020). Thus, the non-acidified samples were used for biological analysis, and the acidified samples were used for the analysis of heavy metals. The same sampling procedures were applied on the dump site to collect the leachate samples for elemental analysis and leachate samples for heavy metal analysis. In addition, the sampling procedures for water and leachate were performed throughout both the wet and dry seasons.

A total of six (6) water samples for elemental analysis and six (6) water samples for heavy metal analysis were obtained from the three designated stations along the stream (US, MS, and DS), constituting the first batch of twelve (12) water samples for the wet season (May 2023). Additionally, two bottles of leachate samples for elemental analysis and two leachate samples for heavy metal analysis were also collected during this wet season. The same number of water and leachate samples were also collected in the dry season (August 2023) to examine the effects of seasons on the physical and chemical characteristics, and heavy metal levels in the water and leachate. To minimize temperature fluctuations, water and leachate sampling was conducted in the morning, between the hours of 8:00 and 9:30. After proper labeling, all samples were stored at 4°C in a sampling cooler box and transported to the laboratory at the University of Rwanda's College of Sciences and Technology.

3.4.5. Assessment of the physical, chemical, and bacterial properties of water and leachate

Both the water and leachate samples underwent analysis to evaluate their physicochemical characteristics as well as their bacterial composition. The water physicochemical properties and heavy metals analyzed include total dissolved solids, pH, total organic carbon, electrical conductivity, lead, chromium and cadmium, dissolved oxygen, and chemical oxygen demand. The analyzed bacteria were total viable count, total *coliforms*, and *Escherichia coli*.

Water pH, EC, TDS, and DO was measured using a Ohaus ST10 pH meter pen, an Orion star A 222 conductivity meter, OHAUS TDS meter, and Ohaus 400D dissolved oxygen meter respectively. Chemical Oxygen Demand was determined by a standard HACH procedure as described by (Lange, 2019), where a DR 6000 spectrophotometer and DRB 200 digester were used, and a volume of two milliliters of sample were put in the COD vial and digested at 150 °c for two hours.

Atomic absorption spectrometry (AAS) was utilized to quantify the concentration of cadmium (Cd) in both water and leachate samples, employing an air-acetylene flame, according to ISO 5961. At a wavelength of 228,8 nm, the acidified sample was aspirated into the flame, and the cadmium concentration was measured. According to Silva et al. (1996)), this method is applicable to potable, fresh and saline waters and effluents where After ammonium pyrrolidine dithiocarbomate (APDC) and cadmium have been chelated, and extracted into chloroform, the extract was evaporated, and treated with nitric acid to destroy organic matter. The residue was then dissolved in hydrochloric acid. By aspirating to AAS, the final concentration of cadmium in the resultant solution was measured. Finally, the cadmium concentration was determined using the following formula:

Cadmium (mg/l) = $\frac{C \ x \ 100}{V}$

Whereby V = volume of original sample taken in 9.1 in 1 ml

C = Cadmium concentration from the curve.

Lead (Pb) concentration in water and leachate samples was examined utilizing AAS with an air-acetylene flame, according to ISO 8288. At a wavelength of 283.3 nm, the acidified sample was aspirated into the flame, and the lead concentration was then measured. It was reported in the study conducted by Silva et al. (1996) that during this test lead is chelated using ammonium pyrrolidine dithiocarbomate (APDC) and then extracted into chloroform. The extract is subsequently evaporated, treated with nitric acid to destroy the organic matter, and the residue is dissolved in hydrochloric acid. The lead concentration in the resulting mixture is finally assessed by aspirating to AAS. The following formula was the used to get concentration of lead.

Lead (mg/l) $= \frac{C \ x\ 100}{V}$

Whereby V = volume of original sample taken in 9.1 in 1 ml

C = Lead concentration from the curve.

The method used to determine the amount of chromium was flame atomic absorption spectrometry at a wavelength of 357,9 nm, according to ISO 9174, clause 3. This ISO 9174 show that by reference to the calibration graph obtained, the concentration of chromium can be calculated based on the absorbance of the test portion and of the blank solution as follow:

$$P = \frac{(As - Aso) * Vw}{Vp * b}$$

where:

p is the chromium concentration of the sample, in milligrams per litre

As is the absorbance of the test portion

Aso is the absorbance of the blank

Vp is the volume of the acidified sample in millilitres

b is the sensitivity (slope of the calibration graph) in litres per milligram

Vw is the volume of the test portion in millilitres

It should be noted that the chromium concentration results of this study fell within the concentration range of 0.5 mg/l to 20 mg/l, which is the range in which this method can be used to analyze water and waste water as recommended by the ISO 9174.

Total coliform analysis in water and leachate samples was done using a membrane Filtration method as described by Forster and Pinedo (2015). Samples were filtered using the Millipore membrane filters with pore size 0.45µm and 47mm diameter with a vacuum speed of 5 to 15mmHg. Microbes were retained on the surface of the membranes. The membrane filters were then placed on mEndo agar LES growth medium, and incubated at 35°C for 24 hours. The colonies formed were counted using a colony counter. While the non-coliforms displayed the red colonies, coliforms showed a golden sheen or dark red colonies, *Escherichia coli* developed colonies with a metallic sheen. The total coliforms and *Escherichia coli* results were presented as colony Forming Units per 100ml (cfu/100 ml). The densities of total coliform and *Escherichia coli* were calculated using the following formula:

$$\text{Colonies/ 100 ml} = \frac{colonies\ counted\ X\ 100\ ml}{sample\ volume\ (ml)}$$

ISO 6222 test method was used in the enumeration of culturable micro-organisms (total viable count). One milliliter of each sample was placed in the Petri dish and mixed with the agar culture medium carefully by the gentle rotation, and then allowed to set. After that, the plate was incubated for forty-four hours at 37 °C to create the colonies. The colonies formed on each plate were counted and utilized for the calculation of the number of colony-forming units presented in 1 ml of sample**.** This means that results of total viable count were documented as the number of colony-forming units per milliliter (cfu/ml).

3.4.6 Evaluation of the diversity of plant communities around Cyuve dump site

The present study evaluated plant species of the above designated distances that were used for collecting soil samples downstream from the dump site. Generally, the first distance extended from the edge of the dumpsite up to forty meters (0 - 40), the second from forty meters to eighty meters (40 – 80), and the third from eighty meters to one hundred and twenty meters (80 - 120) distance, aiming to assess potential variations in plant communities across those specified distances (intervals). Systematic quadrat, observation, and recording methods were used to collect plant species data. Since uncontrolled dump sites always drain the leachate in a downstream direction, the study considered the same control site that was used for soil sample comparison, which was located at one hundred meters (100 m) upstream the dump site. This control site location was selected following the recommendations of U.S. Environmental Protection Agency as already described above (U.S. EPA, 2020).

Plant species data were collected following a 40m line transect to record the identified plant species at three different locations on each designated distance (interval) as shown in the Figure 3.5. To collect data on various plant species, this study systematically placed quadrats of different sizes at designated intervals downstream of the dumpsite and at a control site. The number of plant species within these quadrats was then counted. Specifically, a quadrat size of 1x1 meter was utilized for gathering herbaceous plant species, while a 5x5 meter quadrat size was employed for collecting shrub, and a 10x10 meter quadrat size was used for tree species

40m

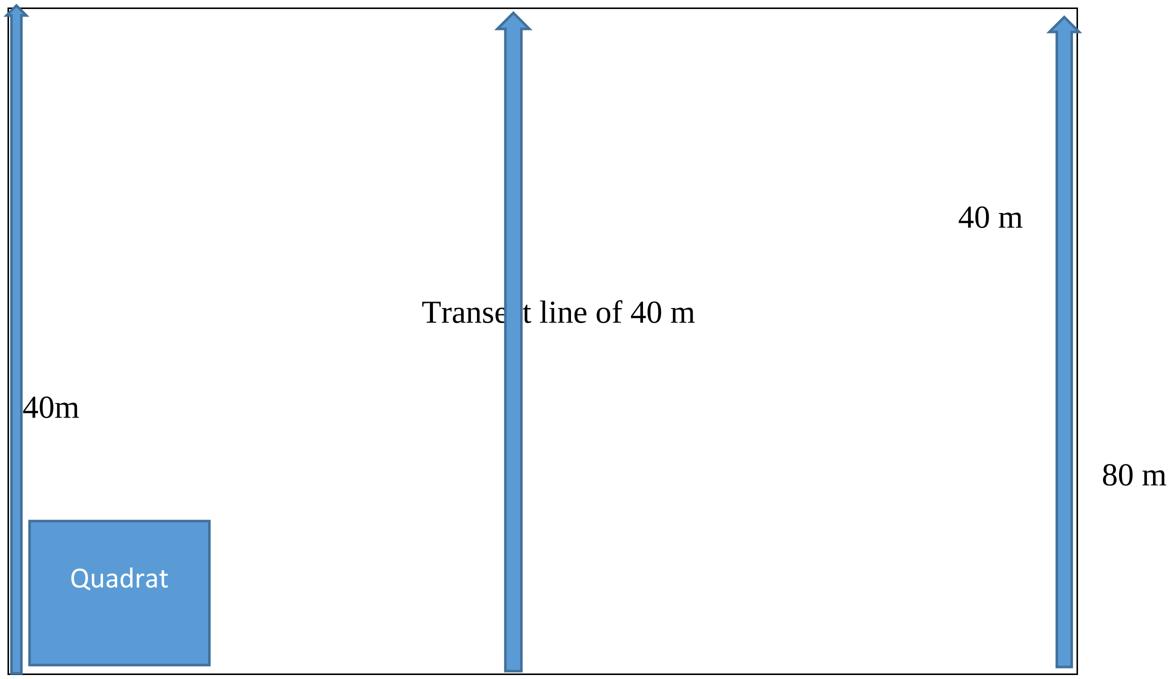

Figure 3.5: Illustration of transect lines for plant species sampling on each distance

Quadrats were systematically located across each distance along three fixed line transects on each distance as illustrated above. These transects extended the length of the sampled distance. To gather herbaceous data, quadrats measuring 1 x 1 meters were placed every 8 meters along each transect line, resulting in a total of five quadrats per line. For shrub data, quadrats measuring 5 x 5 meters were positioned every 5 meters along the transect line, yielding a total of four quadrats per line. Finally, quadrats measuring 10 x 10 meters were situated every 5 meters along the transect line to collect data on trees, resulting in three quadrats per line. This replication was conducted to ensure an accurate representation of each plant species at every investigated distance and control site. It should be noted that the quadrats were laid along the fixed transect lines at each sampled distance downstream the dump site and control site.

Therefore, the study acquired fifteen quadrats for herbaceous plant species, twelve quadrats for shrubs species, and nine quadrats for tree species at each sampled distance and control site. Plant species sampling took place in May 2023, as Rwanda's rainy season was heading to an end. As suggested by Andrade *et al.* (2019), this period marks the optimal time for sampling herbaceous plants due to increased likelihood of flowering and fruiting, thereby reducing the risk of missing plant species as compared to dry seasons. Flowers serve as effective identification for different species during this time.

A comprehensive enumeration **(Counting)** and identification of plants were conducted on each distance, and control site. The assistance in species identification on the field, was provided by a master student in Agroforestry from the University of Rwanda who under graduated in botany and conservation from the same university. Any species that couldn't be identified on-site were systematically coded and transported to the Huye district, where the herbarium at the University of Rwanda facilitated the proper identification. During the fieldwork, the number of individuals belonging to each species, as well as the total number of plants at each distance, were recorded using various sizes of quadrats.

3.5 Data analysis

The study employed RStudio version 4.3.1 for data analysis. Mean values of soil and water parameters were compared utilizing a one-way analysis of variance (ANOVA) and a least significant difference (LSD) test. This comparison was conducted with regards to specified distances, soil depths, and sampled stations along the stream. Furthermore, a t-test was performed to evaluate the impact of seasons on the studied soil and water parameters. Significance of differences in mean values was determined based on calculated p-values, considering values < 0.05 as statistically significant. The study also measured plant diversity, where Simpson diversity and Sorensen similarity indices were employed to ascertain and compare the diversity of plant species across the various distances investigated and the control site. The study used the Gini-Simpson index, also referred to as the complement (1-D) of Simpson's diversity index, serves as a quantitative measure indicating both the variety of species present and the evenness of their distribution, based on Simpson's work in 1949.

Simpson’s diversity index D was computed as described in the study conducted by Ariyo (2020)

$$D= \frac{\sum ni(ni-1)}{N(N-1)}$$

Where D is the Simpson’s diversity index,

ni is the total number of each individual species

N is the total number of all species

Gini-Simpson index = 1-D

In order to determine the interspecific relationship among the species in plant communities, the study also computed the similarity index. The similarity of plant species across the investigated distances and the control site was determined using the Sorensen similarity index. This comparison involved assessing the similarity between the following pairs:

i. Distances of 0 – 40 m and 40 – 80 m
ii. Distances of 0 – 40 m and 80 – 120 m
iii. Distances of 0 – 40 m and the control site
iv. Distances of 40 – 80 m and 80 - 120 m
v. Distances of 40 – 80 m and the control site
vi. Distances of 80 - 120 m and the control site.

Each pair was evaluated to measure the extent of similarity in plant species composition between the specified distances. The Sorensen’s species similarity index was computed as described by the study conducted by Nath *et al*. (2005).

Sorenson's species similarity index = $\left(\frac{2c}{(a+b)}\right) * 100$

Where C is the number the common species in the compared communities, a is the total number of species observed in community 1, and b is the total number of species observed in community 2.

CHAPTER FOUR

RESULTS

4.1 Introduction

Soil layers serve as both a pollutant sink and a filter for drinking water. As described in the study conducted by Samuroupa and Ogbeibu (2020), plants are currently exposed to high concentrations and high mobility of contaminants in the soil, which hinders the growth and development of those plants. The purpose of this study was to determine the Cyuve waste dump site on plant diversity, soil and water physicochemical properties. The present study assessed the variations of the soil and water properties among the different investigated locations. For the effective statistical analysis, the recorded values for the density of total viable count, total coliforms, and *Escherichia coli* in leachate and water samples were converted into log values (logarithm to the base 10).

4.2 Determination seasonal effects on the properties of the leachate

A t-test was conducted to compare the mean values of the physicochemical characteristics, bacterial density, and heavy metal concentration in the sampled leachate during two distinct seasons (wet and dry). The outcomes from the t-test revealed p-values of greater than 0.05 for pH (0.2008), TOC (0.176), EC (0.1791), TDS (0.1214), Pb (0.7285), Cr (0.2203), Cd (0.7879), DO (0.5011), COD (0.06985), TVC (0.0958), Total coliforms (0.3656), *Escherichia coli* (0.6886) indicating that they were not significantly varying among the seasons (Table 4.1).

Table 4.1: Mean values of the physicochemical parameters of leachate in two different seasons

Table 4. 1: Mean values of the physicochemical parameters of leachate in two different seasons

Seasons	**pH**	**TOC**	**Ec**	**TDS**	**Pb**	**Cr**	**Cd**	**DO**	**COD**	**TVC (Log_{10})**	**T.coliforms (Log_{10})**	**E.coli (Log_{10})**
Wet	8.46 [a]	4607.5 [a]	3268.7 [a]	4299.8 [a]	4.7 [a]	7.5 [a]	3.07 [a]	3.45 [a]	2620.5 [a]	3.354 [a]	5.275 [a]	2.676[a]
Dry	8.655 [a]	4086 [a]	3011 [a]	3905.5 [a]	4.2 [a]	8.8 [a]	3.365 [a]	3.34 [a]	2082.5 [a]	3.603 [a]	5.354 [a]	2.746[a]

pH= acidity or alkalinity of the soil, TOC= Total organic carbon (mg/l), *EC= Electrical conductivity* (µS/cm), *TDS= total dissolved solids (ppm), DO= dissolved Oxygen* (mg/l), *COD= chemical oxygen demand* (mg/l), *Pb= Lead* (mg/l), *Cr= Chromium*(mg/l), *Cd= Cadmium*(mg/l), *TVC= Total viable count (cfu/1ml), T.coliforms= Total coliforms (cfu/100ml), E.coli= Escherichia coli (cfu/100ml), ppm = parts per million, mg/l = milligram per liter of water, cfu/1ml= colony-forming units per 1 milliliter, cfu/100ml= colony-forming units per 100 milliliters, Mean values followed by the same superscript letter are not significantly different (at 5%)*

4.3 Determination of the effects of leachate on soil downstream the Cyuve dumpsite

4.3.1 Description of analysis of variance results for soil parameters at varying distances downstream the dumpsite

An analysis of variance (ANOVA) was conducted to compare the mean values of soil physicochemical parameters across varying distances (0 – 40 cm, 40 – 80 cm, 80 -120 cm) from the Cyuve dumpsite, in addition to a control site. The results, as indicated by the asterisks in Table 4.2, demonstrated significant differences ($p < 0.001$) in the mean values of soil electrical conductivity (EC), sand content, silt content, and Chromium (Cr) concentration. Moreover, total dissolved solids (TDS) and lead (Pb) mean values also displayed significant differences ($p < 0.01$), while soil cation exchange capacity (CEC) demonstrated a significant difference ($p < 0.05$) across the measured distances during wet seasons. However, results indicated that mean values of soil organic matter (OM), clay content, and Cadmium (Cd) did not show significant differences ($P > 0.05$) across the measured distances (Table 4.2). These results indicated that although the closeness to the dumpsite was the key for some soil parameters to be affected by leachate, others did not significantly vary over the tested distances.

During the dry season, results revealed significant differences ($p < 0.001$) in the mean values of soil electrical conductivity (EC), sand content, lead (Pb) concentration, and chromium (Cr) concentration among the investigated distances and the control site. Additionally, total dissolved solids (TDS) and clay content mean values also exhibited significant differences ($p < 0.01$), while soil pH mean values showed a significant difference ($p < 0.05$) across the mentioned distances and control site. However, the mean values of soil organic matter (OM), cation exchange capacity (CEC), silt content, and Cadmium (Cd) concentration did not demonstrate significant differences ($p > 0.05$) across the varying distances and control site. (Table 4.2). These results suggest that the proximity to the Cyuve dumpsite significantly influences certain soil physicochemical parameters, such as electrical conductivity (EC), heavy metal concentrations (Chromium and lead), and total dissolved solids (TDS), particularly during both wet and dry seasons, while other parameters like organic matter (OM) and cadmium (Cd) remain relatively unaffected by the distance from the dumpsite.

Table 4. 2: Analysis of variance test results for soil parameters at varying distances downstream the cyuve dumpsite

Seasons	source of variation	df	pH	OM	EC	TDS	CEC	Sand	Silt	Clay	Pb	Cr	Cd
Wet season	**Distance**	3	1.277***	1.698	31113***	929.7**	129.13*	173.56***	90.83***	13.944	0.7036**	29.872***	0.0945
	Residual	20	0.139	1.901	485	118.1	30.89	15.53	9.77	4.633	0.1201	0.152	0.0679
	mean		6.983	3.8833	165.26	53.448	20.611	49.833	27.42	22.75	0.4576	3.2558	0.2625
	p-value		0.017	0.77	$4.51e^{-7}$	0.007	0.371	0.0001	0.987	0.0079	0.00005	$4.00e^{-11}$	0.223
	LSD at 5%		0.4491	1.6605	26.517	13.087	6.6931	4.7466	3.764	2.5923		0.4688	
Dry season	**Distance**	3	0.7306*	0.4885	14444***	944.1**	50.37	138***	0.444	126.4**	2.2245***	35.33***	0.1864
	Residual	20	0.1707	1.3091	561	178.3	45.67	11.4	9.8	24.2	0.1608	0.47	0.1129
	mean		7.1152	3.1413	135.4	49.529	19.726	49.5	24.67	25.833	0.6943	3.8113	0.3
	p-value		0.001	0.462	$1.90e^{-10}$	0.001	0.018	0.00016	0.0004	0.054	0.0061	$5.10e^{-15}$	0.293
	LSD at 5%		0.4976	1.3779	28.522	16.08	8.1391	4.0663	3.77	5.9245	-	0.8241	-

Df=degree of freedom, m.s= mean of squares, pH= acidity or alkalinity of the soil, OM= organic matter (%), EC= Electrical conductivity (µS/cm), *TDS= total dissolved solids* (ppm), *CEC= Cation exchange capacity* (meq/100g), *Pb= Lead* (mgkg-1), *Cr= Chromium* (mgkg-1), *Cd= Cadmium* (mgkg-1), *ppm = parts per million, % = percentage, mgkg^{-1} = milligram per kilogram of soil, LSD= Least significant difference, *= Showed a significant difference at 5% level of significance, **= Showed a significant difference at 1% level of significance, ***= Showed a significant difference at 0.1% level of significance*

4.3.2 Description of least significant difference test results of soil parameters downstream the dumpsite

The Least Significant Difference (LSD) test at a significance level of 5% was employed to assess variations in the mean values of the measured physicochemical properties of the soil among the investigated distances and the control site. Results indicated that soil pH mean values at the investigated distances around the dump site did not exhibit significant differences ($P>0.05$). However, they were all significantly higher than the soil pH mean values obtained from the control site for both the wet and dry seasons (Table 4.3). Additionally, results revealed that the mean values of organic matter (OM) showed no significant differences across the investigated distances and the control site for both wet and dry seasons (Table 4.3).

According to the obtained results, the mean values of electrical conductivity (EC) recorded at distances 0-40 meters and 40-80 meters from the dump site during the rainy season were significantly higher ($p<0.05$) compared to those at 80-120 meters and the control site (Table 4.3). During the dry season, the closest distance to the dump site (0-40 meters) exhibited the highest EC mean value while the farthest distance (80-120 m) recorded the lowest EC mean value. Distances spanning from 0-40 meters and 80-80 meters downstream the dump site recorded significantly higher mean values of electrical conductivity (EC) compared to the control site across both wet and dry seasons (Table 4.3). Moreover, results indicated that the mean values of total dissolved solids (TDS) recorded at distances 0-40 meters and 40-80 meters from the dump site were significantly higher ($p<0.05$) compared to those at 80- 120 meters and the control site for both seasons (Table 4.3).

Based on the results gathered, during the rainy season, the mean values of soil cation exchange capacity (CEC) recorded at 0-40 meters were significantly higher ($p<0.05$) than those observed at distances of 40-80 meters, 80-120 meters, and the control site. However, during the dry season, results indicated that CEC mean values obtained from the investigated distances and the control site did not show significant differences (Table 4.3). Furthermore, the results revealed that the investigated distances (0-40 meters, 40-80 meters, 80-120 meters) downstream the dump site recorded lower percentages of sand compared to the control site for both seasons. Additionally, for both seasons, results showed that the highest mean percentage value of silt was recorded at the distance of 0-40 meters, closest to the dump site, while the lowest mean value was recorded at the farthest distance of 80-120 meters (Table 4.3).

Based on the results obtained, for both the wet and dry seasons, the dump site's closest distance (0-40 m) showed significantly higher mean values of clay percentage in soil, whereas the farthest distance (80-120 m) showed the lowest clay percentage mean values. Moreover, results further indicated that the mean values of clay percentage recorded from the investigated distances around the dumping site were significantly higher than the clay percentage values recorded at the control site (Table 4.3). Additionally, the results revealed that, for both seasons, the dumpsite's closest distance (0-40 m) recorded significantly higher mean values of lead (Pb) and chromium (Cr) concentrations compared to the other investigated distances (40-80 m and 80-120 m) downstream the dumping site. However, all the investigated distances around the dumping site recorded significantly higher concentration mean values of lead (Pb) and chromium (Cr) than the control site (Table 4.3).

Furthermore, during the rainy season, results showed no significant difference in cadmium concentrations across the investigated distances downstream the dumpsite. However, during the dry season, the highest cadmium concentration mean value was recorded at the distance (0-40 m) closest to the dump site, while the lowest was obtained from the farthest distance (80-120 m), although all mean values of cadmium concentration recorded downstream the dump site were significantly higher than the value obtained from the control site (Table 4.3). The findings suggest that proximity to the dump site significantly influences the soil's physicochemical properties and heavy metal concentrations, suggesting that contamination from waste dumping impacts soil quality and environmental health, especially in the areas nearest to the dump site.

Table 4. 3: Mean values of the soil physicochemical parameters and heavy metals concentrations at different distances downstream the Cyuve dump site

Seasons	Distance (m)	pH	OM	EC	TDS	CEC	Sand	Silt	Clay	Pb	Cr	Cd
Wet	**0 - 40**	7.3[a]	4.45[a]	230.72[a]	65.358[a]	27.1[a]	45.6[b]	32.7[a]	21.7[a]	0.88[a]	5.44[a]	0.54[a]
	40 – 80	7.2[a]	4.18[a]	224.28[a]	62.892[a]	20.5[ab]	54[a]	29[b]	17[b]	0.52[ab]	4.55[b]	0.26[a]
	80 -120	7.03[a]	3.71[a]	107.58[b]	44.475[b]	18.7[b]	55.7[a]	25[c]	19.3[ab]	0.19[b]	2.62[c]	0.353[a]
	Control site	6.31[b]	3.23[a]	98.467[b]	41.067[b]	16.2[b]	56[a]	27[bc]	17[b]	0.14[b]	0.41[d]	ND
Dry	**0 - 40**	7.3[a]	3.48[a]	205.37[a]	62.508[a]	22.5[a]	48.4[c]	29[a]	22.6[a]	1.49[a]	6.13[a]	0.79[a]
	40 – 80	7.2[a]	3.19[a]	133.28[b]	57.975[a]	21.8[a]	53[ab]	27.2[a]	19.8[ab]	0.76[b]	5.14[b]	0.453[b]
	80 – 120	7.2[a]	3.12[a]	104.74[c]	39.981[b]	18.0[a]	54[ab]	25.7[a]	20.3[ab]	0.37[bc]	3.38[c]	0.193[b]
	Control site	6.5[b]	2.78[a]	98.22[c]	37.65[b]	16.6[a]	55.2[a]	25.7[a]	19.1[b]	0.2[c]	0.59[d]	0.14[b]

S1= Sub-unit 1, S2= Sub-unit 2, S3= Sub-unit 3, C= control site, pH= acidity or alkalinity of soil, OM= organic matter (%), EC= Electrical conductivity (µS/cm), *TDS= total dissolved solid (ppm), CEC= Cation exchange capacity* (meq/100g), *Pb= Lead* (mgkg-1), *Cr= Chromium* (mgkg-1), *Cd= Cadmium*(mgkg-1), *ppm = parts per million, % = percentage, mgkg*$^{-1}$ *= milligram per kilogram of soil, Mean values followed by the same superscript letter are not significantly different (at 5%), ND= Not detected*

4.3.3 Description of analysis of variance results for soil parameters at varying depths during the wet season

An ANOVA test was undertaken to compare the values of measured parameters across varying depths (0 – 5 cm, 5 – 15 cm, 15 – 30 cm) at different distances downstream from the dumpsite during the rainy season. The findings indicated significant differences ($p < 0.05$) in the mean values of soil pH, EC, TDS, CEC, Pb, Cr, and Cd among the depths of the dumpsite closest distance (0-40 m). However, the mean values of soil OM, sand content, silt content, and clay content did not show significant differences across the depths ($p > 0.05$). On the other hand, these findings indicated that certain soil parameters, such as pH, EC, and heavy metal concentrations, varied significantly with depths at the closest distance from the dumpsite during the wet season. While, other parameters like soil OM content and soil texture did not significantly vary across the measured depths. (Table 4.41).

The findings indicated significant differences ($p < 0.05$) in the values of various soil parameters across different depths at distances ranging from 40 to 80 meters downstream from the dumpsite, as well as at distances of 80 to 120 meters and at the control site (Table 4.4). At the distance of 40 to 80 meters, significant variations were noted in the values of soil pH, OM, EC, TDS, Pb, and Cr across the depths. Whereas parameters such as CEC, sand content, silt content, clay content, and Cd did not show significant variations among the depths.

Similarly, at distances of 80 to 120 meters downstream from the dumpsite, significant variations were noted in the mean values of soil pH, OM, EC, TDS, CEC, sand content, Pb, and Cr across different depths. However, parameters such as silt content, clay content, and Cd did not show significant differences among the depths. Furthermore, Results revealed significant differences in the mean values of soil parameters such as pH, OM, EC, TDS, CEC, Pb, and Cr across different depths at the control site. In contrast, parameters including sand content, silt content, and clay content did not show significant variations among the depths (Table 4.42). These results suggest that soil physicochemical properties, particularly pH, electrical conductivity, and heavy metal concentrations, are influenced by both depth and distance from the dumpsite, indicating that contamination effects are not only confined to the surface but extend through the soil profile, while other parameters like organic matter and texture remain relatively stable across different depths.

Table 4. 4: Analysis of variance test results for soil parameters at varying depths during the wet season

Distance (m)	source of variation	df	pH	OM	EC	TDS	CEC	Sand	Silt	Clay	pb	Cr	Cd
0 - 40	**Depths**	2	0.44*	10.71	2843.5***	710.9***	213.7***	24.67	32.67	0.67	0.72*	0.57**	0.33**
	Residual	3	0.02	1.35	12.70	3.2	0.06	4.00	4.00	4.00	0.03	0.01	0.01
	Mean		7.30	4.45	230.7	65.3583	27.07	42.67	32.67	24.67	0.88	5.44	0.543
	P-value		0.01	0.063	0.001	0.0001	0.00001	0.086	0.0611	0.854	0.01	0.01	0.07
	LSD at 5%		0.40	3.70	11.35	5.674	0.78	6.36	6.36	6.36	0.50	0.38	0.29
40 - 80	**Depths**	2	0.11*	2.84*	1110.5***	225.23**	18.9	42.00	18.00	6.00	0.189**	0.26*	0.0003
	Residual	3	0.01	0.25	2.10	1.63	8.90	14.00	9.33	3.33	0.005	0.03	0.0001
	Mean		7.28	4.15	224.28	62.8917	20.45	49.00	28.00	23.00	0.517	4.55	0.2666
	P-value		0.04	0.04	0.0001	0.0011	0.27	0.19	0.29	0.31	0.007	0.05	0.1470
	LSD at 5%		0.30	1.60	4.58	4.05736	9.49	11.91	9.72	5.81	0.221	0.52	-
80 - 120	**Depths**	2	0.25**	1.23*	207.5*	70.6**	29.67**	34.67*	14.00	4.667	0.066**	0.41**	0.0003
	Residual	3	0.00	0.11	14.23	0.93	0.32	2.67	3.33	3.33	0.001	0.01	0.0001
	Mean		7.04	3.72	107.58	44.48	18.70	52.67	25.00	22.33333	0.193	2.62	0.353
	P-value		0.003	0.04	0.028	0.003	0.002	0.033	0.14	0.37	0.002	0.00	0.147
	LSD at 5%		0.19	1.04	12.01	3.07	1.80	5.20	5.81	5.810325	0.092	0.24	
Control site	**Depths**	2	0.55***	1.55*	598.2*	160.73**	32.59***	14.00	2.00	8.00	0.0001***	0.19*	-
	Residual	3	0.00	0.07	29.80	3.24	0.04	6.00	4.00	7.33	0.000	0.01	-
	Mean		6.31	3.23	98.47	41.07	16.23	55.00	24.00	21.00	0.0167	0.41	-
	P-value		0.00	0.02	0.02	0.005	0.00	0.25	0.65	0.44	0.00	0.03	-
	LSD at 5%		0.15	0.85	17.37	5.73	0.61	7.80	6.36489	8.62	0.000	0.35	-

*pH= acidity or alkalinity of the soil, OM= organic matter (%), EC= Electrical conductivity***(μS/cm)***, TDS= total dissolved solids (ppm), CEC= Cation exchange capacity* (meq/100g)*, Pb= Lead* (mgkg-1)*, Cr= Chromium*(mgkg-1)*, Cd= Cadmium* (mgkg-1)*, ppm = parts per million, % = percentage, mgkg*$^{-1}$ *= milligram per kilogram of soil, letters a and b present the significant difference at 5% level*

4.3.4 Description least significant difference test results of soil parameters at different depths in the wet season

The Least Significant Difference test at a 5% significance level was conducted to assess the variation in the mean values of measured physicochemical soil properties among the depths (0-5 cm, 5-15 cm, 15-30 cm) within each investigated distance, and at the control site in wet season. Results indicated that the highest mean values of certain soil parameters, including pH, OM, EC, and TDS, were recorded at the depth of 0-5 cm, while the lowest mean values were recorded at the depth of 15-30 cm across all investigated distances and the control site (Table 4.5). Moreover, obtained results showed that the mean values of CEC recorded at different depths within the distance of 40-80 meters did not show significant difference. However, results showed that the CEC mean values recorded from the distances of 0-40 meters, 80-120 meters, and the control site were significantly higher at the depth of 0-5 cm, while they were lower at the depth of 15-30 cm (Table 4.5).

Results indicated that both the control site and investigated distances downstream the Cyuve landfill showed the highest values of sand percentage at a depth of 0-5 cm, whereas the lowest values were recorded at a depth of 15-30 cm. Specifically, at the control site and the distance of 0-40 meters, results showed that the highest mean values of silt were recorded at a depth of 5-15 cm, with the lowest at a depth of 15-30 cm. On the other hand, at distances of 40-80 meters and 80-120 meters, the highest silt mean values were observed at a depth of 0-5 cm, whereas the lowest values were recorded at a depth of 15-30 cm (Table 4.5).

Moreover, results showed that from all investigated distances and control site, the highest mean values of clay percentage were recorded at a depth of 15-30 cm, whereas the lowest values were obtained at a depth of 0-5 cm. Additionally, results revealed that all distances recorded the highest concentrations of heavy metals (Pb, Cr, Cd) at a depth of 15-30 cm, while the lowest concentrations were observed at a depth of 0-5 cm. However, It is essential to highlight that heavy metals like lead (Pb) and Cadmium (Cd) were not detected in some of the depths of the control site (Table 4.5). These findings indicate that soil physicochemical properties and heavy metal concentrations vary significantly with depth, suggesting that surface layers (0-5 cm) are more influenced by leachate, while deeper layers (15-30 cm) may accumulate pollutants over time, emphasizing the significance of considering soil depth in assessing the impacts of waste dumpsite on soil.

Table 4. 5: Mean values of the soil physicochemical parameters and heavy metal concentrations at varying depths in the wet season

Distance (m)	Depths (cm)	pH	OM	EC	TDS	CEC	Sand	Silt	Clay	Pb	Cr	Cd
0 - 40	0-5	7.8 ^{a}	5.90 ^{a}	264.3 ^{a}	82.13 ^{a}	32.84 ^{a}	50.00 ^{a}	33.00 ^{ab}	17.00 ^{b}	0.28 ^{c}	4.92 ^{c}	0.35 ^{b}
	5-15	7.3 ^{b}	5.67 ^{a}	238.0 ^{b}	69.00 ^{b}	33.23 ^{a}	45.00 ^{ab}	35.00 ^{a}	20.00 ^{b}	0.89 ^{b}	5.43 ^{b}	0.55 ^{b}
	15-30	6.8 ^{c}	1.78 ^{b}	189.9 ^{c}	44.95 ^{c}	15.13 ^{b}	42.00 ^{b}	30.00 ^{b}	28.00 ^{a}	1.48 ^{a}	5.98 ^{a}	0.73 ^{a}
40 - 80	0-5	7.5 ^{a}	5.26 ^{a}	246.4 ^{a}	73.20 ^{a}	23.00 ^{a}	57.00 ^{a}	32.00 ^{a}	11.00 ^{c}	0.22 ^{c}	4.22 ^{b}	0.2 ^{a}
	5-15	7.3 ^{ab}	4.29 ^{ab}	227.0 ^{b}	63.48 ^{b}	21.31 ^{a}	54.00 ^{a}	28.00 ^{b}	18.00 ^{b}	0.51 ^{b}	4.51 ^{ab}	0.24 ^{a}
	15-30	7.1 ^{b}	2.89 ^{b}	199.5 ^{c}	52.00 ^{c}	17.03 ^{a}	51.00 ^{a}	26.00 ^{b}	2300 ^{a}	0.83 ^{a}	4.93 ^{a}	0.36 ^{a}
80 - 120	0-5	7.4 ^{a}	4.26 ^{a}	108.6 ^{ab}	45.83 ^{b}	20.97 ^{a}	59.00 ^{a}	29.00 ^{a}	12.00 ^{c}	0.35 ^{c}	2.21 ^{c}	0.33 ^{a}
	5-15	6.9 ^{b}	4.08 ^{a}	117.3 ^{a}	49.63 ^{a}	20.87 ^{a}	55.00 ^{a}	25.00 ^{ab}	20.00 ^{b}	0.44 ^{b}	2.55 ^{b}	0.28 ^{a}
	15-30	6.7 ^{c}	2.82 ^{b}	97.0 ^{b}	37.98 ^{c}	14.25 ^{b}	53.00 ^{b}	21.00 ^{b}	26.00 ^{a}	0.5 ^{a}	3.1 ^{a}	0.45 ^{a}
Control	0-5	6.8 ^{a}	4.17 ^{a}	114.8 ^{a}	49.15 ^{a}	20.86 ^{a}	60.00 ^{a}	26.00 ^{b}	14.00 ^{b}	ND	0.17 ^{b}	ND
	5-15	5.8 ^{c}	3.08 ^{b}	100.3 ^{a}	42.63 ^{b}	14.38 ^{b}	55.00 ^{a}	30.00 ^{a}	15.00 ^{b}	0.3 ^{b}	0.31 ^{b}	ND
	15-30	6.3 ^{b}	2.43 ^{b}	80.4 ^{b}	31.43 ^{c}	13.46 ^{c}	53.00 ^{a}	25.00 ^{b}	22.00 ^{a}	0.42 ^{a}	0.765 ^{a}	ND

pH= acidity or alkalinity of the soil, OM= organic matter (%), EC= Electrical conductivity(**µS/cm**)*, TDS= total dissolved solids (ppm), CEC= Cation exchange capacity*(meq/100g)*, Pb= Lead* (mgkg-1)*, Cr= Chromium*(mgkg-1)*, Cd= Cadmium* (mgkg-1)*, ppm = parts per million, % = percentage, mgkg*$^{-1}$ *= milligram per kilogram of soil, Mean values followed by the same superscript letter are not significantly different (at 5%), ND= Not detected*

4.3.5 Description of analysis of variance results for soil parameters at varying depths during the dry season

In the dry season, an ANOVA was conducted to evaluate the average values of the soil parameters measured at different depths (0 – 5 cm, 5 – 15 cm, and 15 – 30 cm) across different distances downstream from the dump site. At the closest distance from the dumpsite (0 – 40 meters), results showed significant differences ($p < 0.05$) in the values of certain soil parameters across the tested depths. Parameters including OM, EC, TDS, CEC, Pb, Cr, and Cd showed significant variations among the depths. However, the mean values of pH, sand content, silt content, and clay content did not show significant differences ($p > 0.05$) among the depths (Table 4.6).

The results for the distance range of 40 to 80 meters from the dump site revealed significant differences ($p < 0.05$) in the values of soil parameters such as organic matter (OM) content, EC, TDS, CEC, Pb, and Cd concentrations across the tested depths. On the other hand, parameters including pH, sand content, silt content, clay content, and chromium (Cr) concentration did not show significant differences ($p > 0.05$) among the depths (Table 4.6). These findings indicated that certain soil parameters, such as OM content, EC, TDS, CEC, Pb, and Cd concentrations, exhibit variations across depths within the specified distance range from the dump site. However, parameters like pH, soil texture (sand, silt, and clay), and Cr concentration did not significantly vary across the tested depths.

The results also revealed that, starting at a distance of 80 up to 120 meters from the dump site, significant variations ($p < 0.05$) were observed in the mean values of certain soil parameters across the tested depths. Parameters such as pH, OM, EC, TDS, CEC, sand content, lead (Pb) concentration, and cadmium (Cd) concentration showed significant differences among the depths, while silt content, clay content, and Cd concentration did not show significant variations ($p > 0.05$)

among the depths (Table 4.6). Moreover, an ANOVA analysis indicated that at the control site, the values of pH, OM, EC, and chromium (Cr) concentration showed significant differences ($p < 0.05$) among the depths of the control site. Whereas, TDS, CEC, sand content, silt content, clay content, Pb and Cd concentration were not significantly different ($p > 0.05$) among the tested depths (Table 4.6). These results demonstrate that, during the dry season, certain parameters significantly changing across depths near the dump site, while others, remain stable, indicating potential influences of seasonal factors on soil contamination dynamics.

Table 4. 6: Analysis of variance test results for soil parameters at varying depths during the dry season

Distance (m)	source of variation	df	pH	OM	EC	TDS	CEC	Sand	Silt	Clay	pb	Cr	Cd
0 - 40	**Depths**	2	0.09	4.38**	2920.5**	486.4**	129.6**	6.00	52.00	56	0.89**	0.72*	0.74*
	Residual	3	0.06	0.09	77.9	5.2	1.11	3.33	114.00	45.33	0.03	0.07	0.03
	Mean		7.27	3.48	205.37	62.5083	21.85	43.00	25.00	32.00	1.49	6.13	0.79
	P-value		0.38	0.01	0.007	0.001	0.001	0.31	0.56	0.40	0.01	0.04	0.01
	LSD at 5%		0.81	0.95	28.09	7.26494	3.36	5.81	19.62	21.4	0.52	0.85	0.55
40 - 80	**Depths**	2	0.36	3.82**	1377.6**	859.4***	218.95*	6.00	0.67	4.67	0.383**	0.71	0.003*
	Residual	3	0.07	0.03	37.1	0.50	20.93	32.67	4.00	54.00	0.006	1.70	0.000
	Mean		7.35	3.19	133.283	57.98	22.51	49.00	24.67	26.33	0.785	5.14	0.453
	P-value		0.09	0.001	0.007	0.00	0.04	0.84	0.85	0.91	0.003	0.69	0.020
	LSD at 5%		0.82	0.55	19.38	2.17	14.56	18.19	6.36	23.39	0.250	4.15	0.043
80 - 120	**Depths**	2	0.38**	2.93***	702.3**	277.3***	33.7***	26.00*	0.67	18.67	0.204**	0.08	1.7E-05**
	Residual	3	0.01	0.01	16.8	0.8	0.06	1.33	0.67	2	0.002	0.12	0.0001
	Mean		7.24	3.12	104.74	37.65	17.96	52.00	24.33	23.67	0.365	3.38	0.1933
	P-value		0.002	0.001	0.006	0.0002	0.0001	0.02	0.46	0.05	0.001	0.57	0.66
	LSD at 5%		0.23	0.36	13.06	2.85	0.77	3.67	2.60	4.50	0.131	1.10	-
Control	**Depths**	2	0.67***	1.72**	369.00*	123.10	32.57	14.00	4.67	8.67	0.00018	0.29*	0.0002
	Residual	3	0.00	0.03	27.90	17.80	5.81	4.00	1.33	1.33	0.00013	0.029	0.0001
	Mean		6.60	2.78	98.22	39.98	16.59	54.00	24.67	21.33	0.03	0.597	0.1466
	P-value		0.0003	0.004	0.03	0.07	0.10	0.16	0.16	0.08	0.41	0.046	0.33
	LSD at 5%		0.14	0.56	16.80	13.43	7.67	6.36	3.67	3.67	-	0.539	-

*pH= acidity or alkalinity of the soil, OM= organic matter (%), EC= Electrical conductivity***(µS/cm)***, TDS= total dissolved solids (ppm), CEC= Cation exchange capacity***(meq/100g)***, Pb= Lead* **(mgkg-1)***, Cr= Chromium***(mgkg-1)***, Cd= Cadmium* **(mgkg-1)***, ppm = parts per million, % = percentage, mgkg^{-1} = milligram per kilogram of soil, letters a and b present the significant difference at 5% level*

4.3.6 Description of least significant difference test results of soil parameters at different depths in the dry season

The LSD test, conducted at a 5% significance level, to ascertain the differences in the physical and chemical characteristics of soil across various depths (0-5 cm, 5-15 cm, 15-30 cm) at each investigated distance downstream from the dump site, as well as at the control site, in the dry season. Results revealed that, during this season, the highest mean values of certain soil parameters, including pH, OM, EC, TDS, sand percentage, and CEC, were recorded at a depth of 0-5 cm. Whilst, the lowest mean values for these parameters were recorded at a depth of 15-30 cm across the investigated distances downstream the Cyuve landfill, and at the control site (Table 4.7).

Results indicated that there was no significant difference in the values of silt percentages between the depths of each investigated distance and the control site ($p > 0.05$) (Table 4.7). Results indicated no significant difference in the mean values of clay percentages between the depths of distances starting at 0-40 m and 40-80 m downstream the dump site ($p > 0.05$). However, at a distance of 80-120 m and the control site, significant differences in clay percentages were observed. Specifically, the highest mean values of clay were recorded at a depth of 15-30 cm, while the lowest mean values were found at a depth of 0-5 cm (Table 4.7). Additionally, according to the obtained results, the higher levels of heavy metals (Pb, Cr, Cd) were obtained at a depth of 15-30 cm across all investigated distances and at the control site. Conversely, the lowest levels of heavy metals were detected at a depth of 0-5 cm across all investigated distances downstream from the dump site (Table 4.7). These findings indicate that heavy metal concentrations tend to accumulate at greater depths, while surface layers exhibit higher values for other soil properties, underscoring the complex interactions between soil characteristics and depth.

Table 4. 7: Mean values of the soil physicochemical parameters and heavy metal concentrations at different depths in the dry season

Distance (m)	Depths (cm)	pH	OM	EC	TDS	CEC	Sand	Silt	Clay	pb	Cr	Cd
0 – 40	0-5	7.5 ^{a}	4.72 ^{a}	242.3 ^{a}	77.6 ^{a}	30.0 ^{a}	52.0 ^{a}	28.0 ^{a}	20.0 ^{a}	0.76 ^{b}	5.6 ^{b}	0.43 ^{b}
	5-15	7.3 ^{a}	3.87 ^{a}	207.9 ^{b}	63.4 ^{b}	21.7 ^{b}	46.0 ^{a}	30.0 ^{a}	24.0 ^{a}	1.6 ^{a}	6.0 ^{ab}	0.84 ^{b}
	15-30	7.1 ^{a}	1.84 ^{b}	166.0 ^{c}	46.5 ^{c}	13.9 ^{c}	48.0 ^{a}	28.0 ^{a}	24.0 ^{a}	2.06 ^{a}	6.8 ^{a}	1.09 ^{a}
40 - 80	0-5	7.62 ^{a}	4.50 ^{a}	161.5 ^{a}	74.75 ^{a}	33.7 ^{a}	58.0 ^{a}	27.0 ^{a}	15.0 ^{a}	0.39 ^{c}	5.76 ^{a}	0.12 ^{b}
	5-15	7.57 ^{a}	3.33 ^{b}	128.8 ^{b}	64.38 ^{b}	20.7 ^{ab}	52.0 ^{a}	24.0 ^{a}	24.0 ^{a}	0.71 ^{b}	4.58 ^{a}	0.34 ^{b}
	15-30	6.86 ^{a}	1.74 ^{c}	109.6 ^{b}	34.8 ^{c}	13.1 ^{b}	49.0 ^{a}	31.0 ^{a}	20.0 ^{a}	1.26 ^{a}	5.07 ^{a}	0.9 ^{a}
80 - 120	0-5	7.7 ^{a}	4.15 ^{a}	124.0 ^{a}	51.15 ^{a}	20.9 ^{a}	59.0 ^{a}	24.0 ^{a}	17.0 ^{b}	0.1 ^{c}	3.34 ^{a}	NA
	5-15	7.1 ^{b}	3.43 ^{b}	103.7 ^{b}	32.3 ^{b}	19.7 ^{b}	54.0 ^{a}	27.0 ^{a}	19.0 ^{ab}	0.28 ^{b}	3.21 ^{a}	0.26 ^{a}
	15-30	6.9 ^{c}	1.79 ^{c}	86.6 ^{c}	29.5 ^{b}	13.3 ^{c}	49.0 ^{b}	26.0 ^{a}	25.0 ^{a}	0.72 ^{a}	3.60 ^{a}	0.32 ^{a}
Control site	0-5	6.8 ^{b}	3.59 ^{a}	112.7 ^{a}	48.87 ^{a}	20.4 ^{a}	59.0 ^{a}	25.0 ^{a}	16.0 ^{b}	0.2 ^{a}	0.33 ^{b}	NA
	5-15	7.2 ^{a}	2.99 ^{b}	96.1 ^{ab}	37.05 ^{ab}	16.9 ^{ab}	55.0 ^{a}	24.0 ^{a}	21.0 ^{ab}	0.23 ^{a}	0.44 ^{b}	0.19 ^{a}
	15-30	6.6 ^{c}	1.77 ^{c}	85.8 ^{b}	34.03 ^{b}	12.4 ^{b}	51.0 ^{a}	29.0 ^{a}	20.0 ^{a}	0.44 ^{a}	1.03 ^{a}	0.25 ^{a}

pH= acidity or alkalinity of the soil, OM= organic matter (%), EC= Electrical conductivity(µS/cm), TDS= total dissolved solids (ppm), CEC= Cation exchange capacity(meq/100g), Pb= Lead (mgkg-1), Cr= Chromium(mgkg-1), Cd= Cadmium (mgkg-1), ppm = parts per million, % = percentage, mgkg^{-1} = milligram per kilogram of soil, Mean values followed by the same superscript letter are not significantly different (at 5%)

4.3.7 Description of results of seasonal variation on soil parameters

A t-test was used to assess the values of the measured soil physicochemical properties and heav y metals between the two different seasons (wet and dry). The results of this t-test indicated that s e asons did not have a significant effect on many of the measured soil parameters across all invest i gated distances and the control site. For instance, the obtained p-values of 0.8816, 0.7195, 0.347 2, and 0.3461 for soil pH in the distances of 0-40 m, 40-80 m, 80-120 m, and the control site, resp ectively, were greater than 0.05 ($p > 0.05$). This indicates that seasons did not significantly affect the soil pH within these distances (Table 4.8).

The obtained p-values of 0.3902, 0.1937, 0.3002, and 0.374 for organic matter (OM) in the soil o f the distances of 0-40 m, 40-80 m, 80-120 m, and the control site, respectively, were greater tha n 0.05 ($p > 0.05$). This indicates that seasons did not have a significant effect on the soil organic matter (OM) content within these distances (Table 4.8). Furthermore, the obtained p-values of 0.2 301, 0.7311, and 0.9771 for electrical conductivity (EC) in the soil at distances of 0-40 m, 80-120 m, and the control site, respectively, were greater than 0.05 ($p > 0.05$). This indicates that soil ele ctrical conductivity in these distances was not significantly affected by the seasons. However, the obtained p-value (4.103e-05) at the distance of 40-80 m was less than 0.05 ($p < 0.05$), indicating t hat seasons significantly affected the electrical conductivity within this distance (Table 4.8).

The obtained p-values of 0.7577, 0.5807, 0.1985, and 0.8177 for total dissolved solids (TDS) in t he soil at distances of 0-40 m, 40-80 m, 80-120 m, and the control site, respectively, were greater than 0.05 ($p > 0.05$). This indicates that total dissolved solids in the soil were not significantly aff ected by seasons (Table 4.8). Moreover, the obtained p-values of 0.3034, 0.6511, 0.7284, and 0.8 747 for soil cation exchange capacity (CEC) at distances of 0-40 m, 40-80 m, 80-120 m, and the control site, respectively, were greater than 0.05 ($p > 0.05$). This indicates that soil CEC was not significantly affected by seasons (Table 4.8).

The obtained p-values of 0.8464, 0.6511, 0.7284, and 0.5679 for sand percentage in the distances of 0-40 m, 40-80 m, 80-120 m, and the control site, respectively, were greater than 0.05 ($p > 0.05$). This indicates that the soil sand content was not significantly affected by seasons (Table 4.8). Furthermore, the obtained p-values of 0.07654, 0.5911, and 0.5156 for silt percentage in the soil at distances of 40-80 m, 80-120 m, and the control site, respectively, were greater than 0.05 ($p > 0.05$). This indicates that silt percentages in the soil was not significantly affected by seasons. However, the obtained p-value of 0.02513 for the distance of 0-40 m was less than 0.05 ($p < 0.05$), indicating that seasons significantly affected the silt content in soil at this distance (Table 4.8).

The obtained p-values of 0.05084, 0.2085, 0.2398, and 0.8178 for clay percentage in the soil at distances of 0-40 m, 40-80 m, 80-120 m, and the control site, respectively, were greater than 0.05 ($p > 0.05$). This indicates that the clay content in soil was not significantly affected by seasons (Table 4.8). Moreover, the p-values of 0.1025, 0.2352, 0.3811, and 0.08363 for lead (Pb) concentrations in the soil at distances of 0-40 m, 40-80 m, 80-120 m, and the control site, respectively, were greater than 0.05 ($p > 0.05$. This indicates that the concentration of lead (Pb) in the soil was not significantly affected by the seasons, although there was a slight increase in these heavy metals during the dry season (Table 4.8).

The obtained p-values of 0.04858 and 0.005384 for chromium (Cr) concentration in the soil at distances of 0-40 m and 80-120 m, respectively, were less than 0.05, indicating that chromium concentration in the soil at these distances was significantly affected by the seasons. However, the obt ained p-values of 0.2769 and 0.3594 from the distance of 40-80 m and the control site, respective ly, were greater than 0.05, indicating that the concentration of this heavy metal (Cr) was not signi ficantly affected by seasons, although the study revealed a slight increase in these heavy metal du ring the dry season (Table 4.8).

Moreover, the obtained p-values of 0.6141, 0.1717, and 0.7971 for the concentration of cadmium (Cd) in the soil at distances of 0-40 m, 40-80 m, and 80-120 m, respectively, were greater than 0. 05, indicating that cadmium (Cd) concentration was not significantly affected by seasons, althoug h the present study indicated a rise in cadmium concentration during the dry season compared to t he wet season (Table 4.8). These results suggest that while most soil physicochemical properties and heavy metal concentrations remain stable across seasons, specific parameters like electrical c onductivity and chromium concentration exhibit significant seasonal variations, highlighting the i nfluence weather conditions.

Table 4. 8: Mean values of the soil physicochemical parameters and heavy metal concentration in different seasons

Distance (m)	Seasons	pH	OM	EC	TDS	CEC	Sand	Silt	Clay	pb	Cr	Cd
0 - 40	Wet	7.3 [a]	4.45 [a]	230.72 [a]	65.36 [a]	27.07 [a]	42.67 [a]	32.67 [a]	24.67 [a]	0.88 [a]	5.44 [b]	0.54 [a]
	Dry	7.3 [a]	3.48 [a]	205.37 [a]	62.51 [a]	21.85 [a]	43.00 [a]	25.00 [b]	32.00 [a]	1.49 [a]	6.13 [a]	0.79 [a]
40 - 80	Wet	7.3 [a]	4.15 [a]	224.28 [a]	62.89 [a]	20.45 [a]	49.00 [a]	28.00 [a]	23.00 [a]	0.52 [a]	4.55 [a]	0.25 [a]
	Dry	7.4 [a]	3.19 [a]	133.28 [b]	57.98 [a]	22.51 [a]	49.00 [a]	24.67 [a]	26.33 [a]	0.79 [a]	5.14 [a]	0.353 [a]
80 - 120	Wet	7.0 [a]	3.72 [a]	107.58 [a]	44.48 [a]	18.70 [a]	52.67 [a]	25.00 [a]	22.33 [a]	0.19 [a]	2.62 [b]	0.353 [a]
	Dry	7.2 [a]	3.12 [a]	104.74 [a]	37.65 [a]	17.96 [a]	52.00 [a]	24.33 [a]	23.67 [a]	0.37 [a]	3.38 [a]	0.1933 [a]
Control site	Wet	6.3 [a]	3.23 [a]	98.47 [a]	41.07 [a]	16.23 [a]	55.00 [a]	24.00 [a]	21.00 [a]	0.02 [a]	0.41 [a]	-
	Dry	6.6 [a]	2.78 [a]	98.22 [a]	39.98 [a]	16.59 [a]	54.00 [a]	24.67 [a]	21.33 [a]	0.03 [a]	0.60 [a]	-

pH= acidity or alkalinity of the soil, OM= organic matter (%), EC= Electrical conductivity(μS/cm), *TDS= total dissolved solids (ppm), CEC= Cation exchange capacity*(meq/100g), *Pb= Lead* (mgkg-1), *Cr= Chromium*(mgkg-1), *Cd= Cadmium* (mgkg-1), *ppm = parts per million, % = percentage,* $mgkg^{-1}$ *= milligram per kilogram of soil, Mean values followed by the same superscript letter are not significantly different (at 5%)*

4.4 Determination of the effects of Cyuve waste dump site on the water quality of the nearby stream

4.4.1 Description of analysis of variance results of water quality parameters at different stations along the stream

ANOVA test was computed to compare the values of the water physicochemical properties, bacteriological, and heavy metal concentrations among the investigated stations (US, MS, and DS) along the stream. As it is presented by a number of the stars superscripted on the mean of squares in table 4.9, results revealed that during the wet seasons, the mean values of some water parameters such as total dissolved solids (TDS), chemical oxygen demand (COD), Total coliforms and Escherichia coli were significantly different ($p < 0.01$) among the sampled stations along the stream. In addition, the mean values of electrical conductivity (EC), dissolved oxygen (DO), total viable count (TVC), and chromium (Cr) also showed a significant difference ($p < 0.05$). However, mean values of water pH and lead (Pb) concentrations did not show a significant difference ($p > 0.05$) among the stations (Table 4.9).

During the dry season, results showed a significant difference ($p < 0.001$) among the mean values of COD in water of the sampled stations. The mean values of TDS, EC, DO, Total coliforms and Escherichia coli were also revealed significantly different with p value less than 0.01 ($p < 0.01$). Furthermore, the mean values of Pb, Cr, and total viable count (TVC) were significantly different at $P < 0.05$. However, the mean values of pH and cadmium (Cd) did not show a significant difference ($P > 0.05$) across the sampled stations along the stream (Table 4.9). These findings indicate that water quality parameters, especially during the dry season, vary significantly among sampling stations, suggesting that leachate from the Cyuve dump site may be influencing this stream water contamination levels.

Table 4. 9: Analysis of variance test results for water quality parameters along the cyuve dump site nearby water stream

Seasons	source of variation	df	pH	TDS	Ec	DO	COD	Pb	Cd	Cr	TVC	Coliforms	E.coli
Wet season	**Stations**	2	0.1817	6648**	1496*	18.749*	1658**	0.01222	ND	0.05982*	986667*	2.922×10^{9}**	1587950**
	Residual	3	0.0233	74	81	1.062	15	0.00215	ND	0.00315	73333	7.457×10^{7}	21433
	Mean		8.2333	200.45	288.37	7.59	48.032	0.09167	ND	0.21833	933.33	31575	760
	P-value		0.064	0.0021	0.0206	0.022	0.0015	0.095	ND	0.019	0.0384	0.0129	0.0085
	LSD at 5%		0.4861	27.33	28.646	3.28	12.344	0.14756	ND	0.17861	861.81	27481.13	465.914
Dry season	**Stations**		0.1267	8463**	1171.2**	10.371**	2508.1***	0.025*	1.667×10^{-5}	0.4581*	80402*	6.918×10^{9}**	1896617**
	Residual		0.0183	97	10.9	0.194	16.5	0.0021	5×10^{-3}	0.0418	96350	2.138×10^{8}	50550
	Mean		7.9833	230.52	344.55	6.34	52.953	0.1233	0.0133333	0.44333	1318.3	65500	1148.33
	P-value		0.075	0.0022	0.002	0.0045	0.001	0.037	0.667	0.042	0.0463	0.0015	0.0024
	LSD at 5%		0.4309	31.4217	10.504	1.4	12.94	0.1458	NA	0.65091	987.84	46537.04	715.52

US= upstream, MS= middle stream, DS= downstream, pH= acidity or alkalinity of soil, EC= Electrical conductivity (μS/cm), *TDS= total dissolved solids (ppm), DO= dissolved Oxygen* (mg/l), *COD= chemical oxygen demand* (mg/l), *Pb= Lead* (mg/l), *Cr= Chromium*(mg/l) *, Cd= Cadmium*(mg/l) *,TVC= Toatal viable count (cfu/1ml), T.coliforms= Toatal coliforms (cfu/100ml), E.coli= Escherichia coli (cfu/100ml), ppm = parts per million, mg/l = milligram per liter of waterl, cfu/1ml= colony-forming units per 1 milliliter, cfu/100ml= colony-forming units per 100 milliliter, letters a;b;c;d= present the significant difference at 5% level, ND= Not Detectable, RSB= Rwanda Standard Board, WHO= World Health Organization*

4.4.2 Description of least significant difference test results of water quality parameters at different stations along the water stream

The LSD test, conducted at a 5% significance level, aimed to determine the variations in the values of measured physicochemical characteristics and heavy metals among the investigated stations along the dump site nearby stream water. Results revealed that for both seasons, the water pH mean values recorded at the middle stream station (MS) were significantly higher ($p < 0.05$) than the values obtained at the upper stream station (US) and slightly higher than the mean values recorded at the downstream station (DS). However, results indicated that there was no significant difference ($p > 0.05$) among the values of water pH recorded at the upper stream (US) and downstream stream (DS) stations. It is crucial to emphasize that obtained water pH from the investigated stations fell within the ranges established by the Rwanda Standard Board (RSB) and WHO standards for natural potable water (Table 4.10).

Furthermore, results revealed that for both wet and dry seasons, the mean values of water parameters such as TDS, EC, and Chemical Oxygen Demand (COD) were significantly higher at the middle stream (MS) station and lower at the upper stream (US) station. However, the mean values of Dissolved Oxygen (DO) were found to be significantly higher at the upper stream (US) station and lower at the middle stream (MS) station. It is crucial to emphasize that the obtained mean values of TDS and EC fell below the maximum permissible limits established by the Rwanda Standard Board (RSB) and WHO standards for natural potable water. Whilst, the obtained mean values of COD exceeded the maximum permissible limits set by RSB and WHO standards for natural potable water (Table 4.10).

Regarding heavy metals, such as Pb and Cr, the LSD test revealed significantly higher concentration values at the middle stream (MS) station and lower values at the upper stream (US) station for both seasons. However, cadmium was only detected in the stream water samples during the dry season, and results showed no significant difference among the cadmium concentration values recorded at the middle stream (MS) and downstream stations. Notably, concentrations of heavy metals (Pb, Cr, Cd) obtained mainly from the middle stream (MS) exceeded the maximum permissible limits established by RSB and WHO standards for natural potable water (Table 4.10).

Results revealed that for both wet and dry seasons, the mean values of total viable count, total coliforms, and Escherichia coli recorded at the middle stream (MS) station were significantly higher than those recorded at the upper stream (US) station. Conversely, in the wet season, the results indicated a significant difference among the values of total coliforms, and Escherichia coli recorded at the upper stream (US) and downstream (DS) stations (Table 4.10). Likewise, in the dry season, results showed a significant difference among the values of total coliforms and Escherichia coli recorded at the upper stream (US) and downstream (DS) stations.

It is crucial to emphasize that the values of total viable count, total coliforms, and Escherichia coli were converted into log values for proper statistical analysis. In addition, the concentration of total viable count, total coliforms and Escherichia coli that were detected in the Cyuve dump site nearby stream water were revealed to exceed the maximum permissible limits established by RSB and WHO standards for natural potable water (Table 4.10). The observed variations in water quality parameters and heavy metal concentrations highlight the significant impact of the dump site on the stream's ecological health.

Table 4. 10: Mean values of the water quality parameters and heavy metal concentrations with guidelines

Seasons	Stations	pH	TDS	Ec	DO	COD	Pb	Cd	Cr	TVC (log_{10})	Coliform (log_{10})	E.coli (log_{10})
	US	7.9^{b}	139^{c}	270.5^{b}	11.02^{a}	20.305^{c}	0.02^{b}	ND	0.03^{b}	2.239^{b}	2.977^{c}	1.889^{c}
Wet	MS	8.5^{a}	253.35^{a}	319.85^{a}	5.16^{b}	77.78^{a}	0.175^{a}	ND	0.37^{a}	3.196^{a}	4.868^{a}	3.246^{a}
	DS	8.2^{ab}	209^{b}	274.75^{b}	6.58^{b}	46.01^{b}	0.08^{ab}	ND	0.255^{a}	2.998^{ab}	4.278^{ab}	2.626^{ab}
	US	7.7^{b}	165.75^{c}	321^{c}	8.9^{a}	22.02^{c}	0.025^{b}	ND	0.035^{b}	2.389^{b}	4.017^{c}	2.305^{c}
Dry	MS	8.2^{a}	295.85^{a}	369.35^{a}	4.54^{b}	91.58^{a}	0.245^{a}	0.015^{a}	0.97^{a}	3.309^{a}	5.101^{a}	3.329^{a}
	DS	7.9^{ab}	229.95^{b}	343.3^{b}	5.58^{b}	45.26^{b}	0.1^{ab}	0.01^{a}	0.325^{ab}	2.971^{ab}	4.766^{b}	3.035^{ab}
RSB standards		5.5 – 9.5	1500	2500	> 5	< 15	< 0.01	< 0.003	0.05	< 50	ND	ND
WHO standards		6.5 – 8.5	500	1400	≥ 4 - 6	< 5	0.05	0.003	0.1	< 20	ND	ND

US= upstream, MS= middle stream, DS= downstream, pH= acidity or alkalinity of soil, EC= Electrical conductivity (µS/cm), *TDS= total dissolved solids (ppm), DO= dissolved Oxygen* (mg/l), *COD= chemical oxygen demand* (mg/l), *Pb= Lead* (mg/l), *Cr= Chromium*(mg/l) *, Cd= Cadmium*(mg/l) *,TVC= Toatal viable count (cfu/1ml), T.coliforms= Toatal coliforms (cfu/100ml), E.coli= Escherichia coli (cfu/100ml), ppm = parts per million, mg/l = milligram per liter of waterl, cfu/1ml= colony-forming units per 1 milliliter, cfu/100ml= colony-forming units per 100 milliliter, letters a;b;c;d= present the significant difference at 5% level, ND= Not Detectable, RSB= Rwanda Standard Board, WHO= World Health Organization*

Source of the standards presented in the above table: (RSB, 2021), (MININFRA, 2019b), (MININFRA, 2019a), (ICYIMPAYE, 2019), (Sekomo & Bwiza, 2019), (Emmanuel-akerele & Peter, 2020), (Soni & Thomas, 2013).

4.4.3 Determination of seasonal effects on the water quality parameters and heavy metal concentrations

A t-test was conducted to assess the values of the physicochemical and bacteriological properties of surface water, as well as heavy metal concentrations, recorded in two different seasons (wet and dry). The results of this t-test indicated that seasons did not significantly affect some of the measured water quality properties of the investigated stations along the stream. Specifically, the obtained p-values of 0.3959, 0.2726, and 0.1987 when comparing the mean values of pH recorded in the wet and dry seasons at the upper stream (US), middle stream (MS), and downstream (DS) stations, respectively, were greater than 0.05 ($p > 0.05$). This indicates that water pH was not significantly affected seasons (Table 4.11).

The p-value of 0.01151 obtained when comparing the mean values of total dissolved solids (TD S) recorded in wet and dry seasons at the upper stream (US) station was less than 0.05 ($p < 0.05$), indicating that the TDS in water sampled at this station was significantly affected by seasons. Ho wever, the p-values of 0.1733 and 0.2877 obtained at the middle stream (MS) and downstream (D S) stations, respectively, were greater than 0.05 ($p > 0.05$), indicating that TDS in water at these s tations did not significantly affected by seasons.

The p-values of 0.003451 and 0.001667 obtained when comparing the mean values of water ele ctrical conductivity at the upper stream (US) and middle stream (MS) stations, respectively, were less than 0.05 ($p < 0.05$), indicating that the electrical conductivity at these stations was significa ntly affected by seasons. However, the obtained p-value of 0.07933 at the downstream (DS) stati on was greater than 0.05 ($p > 0.05$), indicated that the electrical conductivity at this station was n ot significantly affected by seasons (Table 4.11).

The p-values of 0.318, 0.3825, and 0.1183 obtained when comparing the mean values of dissol ved oxygen (DO) recorded in wet and dry seasons at the upper stream (US), middle stream (MS), and downstream (DS) stations, respectively, were greater than 0.05 ($p > 0.05$). This indicates that the dissolved oxygen (DO) at these stations did not significantly affected by seasons (Table 4.11). likewise, the p-values of 0.464, 0.1452, and 0.8964 obtained when comparing the mean values of Chemical Oxygen Demand (COD) recorded in wet and dry seasons at the upper stream (US), middle stream (MS), and downstream (DS) stations, respectively, were greater than 0.05 ($p > 0.05$). This indicates that seasons did not significantly affect the COD at these stations (Table 4.11).

Additionally, the obtained p-values of 0.7117, 0.4631, and 0.2929, when comparing the mean v alues of lead (Pb) concentrations recorded in wet and dry seasons at the upper stream (US), middl e stream (MS), and downstream (DS) stations, respectively, were greater than 0.05 ($P > 0.05$). Th is indicates that the concentrations of lead at these stations were not significantly affected by the s easons. Similarly, the p-values of 0.7117, 0.2444, and 0.4144 obtained when comparing the mean values of chromium concentrations recorded in wet and dry seasons at the US, MS, and DS statio ns, respectively, were greater than 0.05 ($p > 0.05$). This indicates that seasons did not affect the c oncentration of chromium at these stations (Table 4.11).

The obtained p-values of 0.6842, 0.3713, and 0.8591, obtained when comparing the mean values of total viable count (TVC) recorded in wet and dry seasons at the upper stream (US), middle stream (MS), and downstream (DS) stations, respectively, were greater than 0.05 ($p > 0.05$), indicating that the concentration of total viable count at these stations were not significantly affected by the seasons (Table 4.11). The p-values of 0.0877 and 0.1130, obtained when comparing the mean values of total coliforms recorded at the upper stream (US) and middle stream (MS) stations, were greater than 0.05 ($p > 0.05$), indicating that total coliforms concentrations at these stations were not significantly affected by the seasons. However, the p-value of 0.0050 obtained at the downstream (DS) station was less than 0.05 ($p < 0.05$), indicating that the number of total coliforms at this station was significantly affected by the seasons (Table 4.11).

Furthermore, the p-values of 0.1785 and 0.3318, obtained when comparing the mean values of Escherichia coli recorded in wet and dry seasons at the upper stream (US) and middle stream (MS) stations, were greater than 0.05 ($p > 0.05$), indicating that Escherichia coli concentrations at these stations were not significantly affected by the seasons. However, the obtained p-value of 0.0050 at the downstream (DS) station was less than 0.05 ($p < 0.05$), indicating that total coliforms concentration at this station was significantly affected by the seasons (Table 4.11). The results indicate that while some water quality parameters and heavy metal concentrations remained stable across seasons, others, particularly total dissolved solids and total coliforms at the downstream station, demonstrated significant seasonal variability, reflecting potential weather influences on stream water pollution.

Table 4. 11: Mean values of the water quality parameters and heavy metal concentrations in different seasons

Stations	Seasons	pH	TDS	Ec	DO	COD	Pb	Cr	Cd	TVC (log_{10})	T.coliforms (log_{10})	E.coli (log_{10})
US	Wet	7.95 [a]	139 [b]	270.5 [b]	11.03 [a]	20.31 [a]	0.02 [a]	0.03 [a]	ND	2.239 [a]	2.977 [a]	1.889 [a]
	Dry	7.75 [a]	165.75 [a]	321 [a]	8.9 [a]	22.02 [a]	0.025 [a]	0.035 [a]	ND	2.389 [a]	4.017 [a]	2.305 [a]
MS	Wet	8.55 [a]	253.35 [a]	319.85 [b]	5.16 [a]	77.78 [a]	0.175 [a]	0.37 [a]	-	3.196 [a]	4.868 [a]	3.246 [a]
	Dry	8.25 [a]	295.85 [a]	369.35 [a]	4.54 [a]	91.58 [a]	0.245 [a]	0.97 [a]	-	3.309 [a]	5.101 [a]	3.329 [a]
DS	Wet	8.2 [a]	209 [a]	274.75 [a]	6.58 [a]	46.01 [a]	0.08 [a]	0.255 [a]	-	2.998 [a]	4.278 [b]	2.626 [b]
	Dry	7.95 [a]	229.95 [a]	343.3 [a]	5.58 [a]	45.26 [a]	0.1 [a]	0.325 [a]	-	2.971 [a]	4.766 [a]	3.035 [a]

US= upstream, MS= middle stream, DS= downstream, pH= acidity or alkalinity of soil, EC= Electrical conductivity (µS/cm), *TDS= total dissolved solids (ppm), DO= dissolved Oxygen* (mg/l), *COD= chemical oxygen demand* (mg/l), *Pb= Lead* (mg/l), *Cr= Chromium*(mg/l) *, Cd= Cadmium*(mg/l) *,TVC= Toatal viable count (cfu/1ml), T.coliforms= Toatal coliforms (cfu/100ml), E.coli= Escherichia coli (cfu/100ml), ppm = parts per million, mg/l = milligram per liter of waterl, cfu/1ml= colony-forming units per 1 milliliter, cfu/100ml= colony-forming units per 100 milliliter, letters a;b;c;d= present the significant difference at 5% level, ND= not detected.*

4.5 Plant communities' diversity downstream the Cyuve waste dumping site

4.5.1 Observed plant species

The overall count of plant species observed at each distance and control site is displayed in the table below:

Table 4. 12: Number of observed plant species at varying distances

Distance	Herbs	Shrubs	Trees	Total
0 – 40 m	18	3	2	23
40 – 80 m	22	5	2	21
80 – 120 m	22	2	2	26
Control site	23	4	3	30

The number of plant species identified at various distances downstream from the dumpsite, and at the control site are shown in Table 4.12. It is important to note that while certain plant species were found at all investigated distances downstream of the dumpsite and at the control site, some species were exclusive to specific distances.

Table 4. 13: Observed plant species at different distances from the dump site

Scientific name	Common name	Family	type of plant	0-40 (m)	40-80 (m)	80-120 (m)	Control (m)
Ageratum conyzoides L.	Tropical whiteweed	Asteraceae	herb	+	+	+	+
Amaranthus blitum L.	Purple Amaranth	Amaranthaceae	herb	+	+	+	+
Beta vulgaris L.	Beetroot	Amaranthaceae	herb	+	-	-	-
Bidens pilosa L.	Cobbler’s pegs	Asteraceae	herb	-	+	+	+
Brugmansia suaveolens	Angel's Trumpet	Solanaceae	Shrub	-	+	+	+
Cirsium spinosissimum	Spiniest Thistle	Asteraceae	herb	+	-	-	-
Commelina diffusa Burm.f.	Climbing Dayflower	Commelinaceae	herb	-	+	+	+
Cyperus rotundus L.	Nutgrass	Cyperaceae	herb	+	+	+	+
Digitaria sanguinalis L.	Hairy Crabgrass	Poaceae	Grass	-	+	+	+
Dysphania pumilio	Small Crumbweed	Amaranthaceae	herb	-	+	+	+
Eleusine indica (L.) Gaertn	Crown footgrasss	Poaceae	Grass	+	+	+	+
Erythrina abyssinica	Red-hot Poker Tree	Papilionaceae	Tree	+	-	+	+
Eucalyptus grandis Maidenii	Eucalyptus Gum	Myrtaceae	Tree	-	+	-	+
Eucalyptus maculata	Spotted Gum	Myrtaceae	Tree	+	+	+	+
Euphorbia tirucalli	Pencil Tree	Euphorbiaceae	herb	+	+	-	-
Furcraea foetida	Mauritius Hemp	Asparagaceae	herb	+	+	-	+
Galinsoga quadriradiata	Hairy galinsonga	Asteraceae	herb	-	+	+	+
Guizotia abyssinica	Niger Seed	Asteraceae	herb	-	-	-	+
Hydrocotyle sibthorpioides	Lawn Marshpennywort	Araliaceae	herb	+	+	+	-
Iva xanthiifolia Nutt	Rag sumpweed	Asteraceae	herb	+	-	-	+
Kylinga bulbosa P. Beauv.	Bulb Sedge	Cyperaceae	herb	+	+	+	+
Lepidium didymium L.	Lesser Swine-cress	Brassicaceae	herb	+	+	+	-
Luzula sylvatica (Huds.)	Great Woodrush	Juncaceae	herb	+	+	+	-
Medicago polymorpha L.	Burmedic	Fabaceae	herb	+	+	+	+
Melissa officinalis L.	Lemon Balm	Lamiaceae	herb	-	+	+	+

Mitracarpus hirtus	Shaggy Buttonweed	Rubiaceae	shrub	+	+	+	+
Oxalis corniculata L.	Creeping Woodsorrel	Oxalidaceae	herb	-	+	+	+
Oxalis stricta L.	Yellow Woodsorrel	Oxalidaceae	herb	-	+	+	+
Pavonia urens Cav.	Forest Mallow	Malvaceae	shrub	-	-	-	+
Physalis peruviana L.	Cape Gooseberry	Solanaceae	herb	-	-	-	+
Portulaca oleracea L.	Common Purslane	Portulacaceae	herb	+	+	+	+
Salvia hispanica L.	Chia	Lamiaceae	herb	+	+	+	+
Senna didymobotrya	African Senna	Fabaceae	Shrub	-	+	-	+
Setaria barbata (Lam.) Kunth	bristly foxtail	Poaceae	Grass	+	+	+	+
Sida acuta Burm.f.	Common Wireweed	Malvaceae	herb	+	+	+	+
Solanum americanum Mill.	American Black Nightshade	Solanaceae	herb	+	-	-	-
Tithonia diversifolia (Hemsl.) A. Gray	Mexican Sunflower	Asteraceae	Shrub	+	+	+	+
Total number of present plant species on each site				23	29	26	30

+ = *Presence of plant species in sub-unit*, - = *Plant species is not present the distance*

4.5.2 Evaluation of plant diversity

The Simpson diversity index was selected over the Shannon index because it takes into account both the evenness of the distribution of individuals within a species and the species richness. While the Shannon index can be influenced by sampling design, making comparisons across studies challenging (Andrade *et al*., 2019), the Simpson index, with its scale from 0 to 1, allows for more consistent comparisons across different studies. Moreover, it's crucial to recognize that species diversity serves as a valuable metric for comparing communities impacted by biotic disturbances and for measuring community stability and succession (Ariyo, 2020). As shown in Table 4.13, the plant community diversity index values for the surveyed distances and the control site are 0.91, 0.94, 0.90, and 0.94 for the distances of 0 – 40 m, 40 – 80 m, 80 – 120 m, and the control site respectively.

Table 4. 14: Simpson diversity index for each distance

Scientific name	*Common name*	*Family*	n d1	n(n-1)	n d2	n(n-1)	n d3	n(n-1)	n C	n(n-1)
Ageratum conyzoides L.	Tropical whiteweed	Asteraceae	47	2162	12	132	52	2652	41	1640
Amaranthus blitum L.	Purple Amaranth	Amaranthaceae	19	342	42	1722	7	42	15	210
Beta vulgaris L.	Beetroot	Amaranthaceae	5	20	0	0	0	0	0	0
Bidens pilosa L.	Cobbler’s pegs	Asteraceae	0	0	25	600	81	6480	107	11342
Brugmansia suaveolens	Angel's Trumpet	Solanaceae	0	0	7	42	4	12	4	12
Cirsium spinosissimum	Spiniest Thistle	Asteraceae	2	2	0	0	0	0	0	0
Commelina diffusa Burm.f.	Climbing Dayflower	Commelinaceae	0	0	48	2256	48	2256	56	3080
Cyperus rotundus L.	Nutgrass	Cyperaceae	83	6806	55	2970	9	72	27	702
Digitaria sanguinalis L.	Hairy Crabgrass	Poaceae	0	0	14	182	37	1332	28	756
Dysphania pumilio	Small Crumbweed	Amaranthaceae	0	0	17	272	7	42	26	650
Eleusine indica (L.) Gaertn	Crown footgrasss	Poaceae	36	1260	60	3540	37	1332	101	10100
Erythrina abyssinica	Red-hot Poker Tree	Papilionaceae	4	12	0	0	8	56	3	6
Eucalyptus grandis Maidenii	Eucalyptus Gum	Myrtaceae	9	72	10	90	6	30	10	90
Eucalyptus maculata	Spotted Gum	Myrtaceae	0	0	2	2	0	0	2	2
Euphorbia tirucalli	Pencil Tree	Euphorbiaceae	6	30	7	42	0	0	0	0
Furcraea foetida	Mauritius Hemp	Asparagaceae	11	110	5	20	0	0	2	2
Galinsoga quadriradiata	Hairy galinsonga	Asteraceae	0	0	74	5402	87	7482	58	3306
Guizotia abyssinica	Niger Seed	Asteraceae	0	0	0	0	0	0	5	20
Hydrocotyle sibthorpioides	Lawn Marshpennywort	Araliaceae	14	182	60	3540	3	6	0	0
Iva xanthiifolia Nutt	Rag sumpweed	Asteraceae	2	2	0	0	0	0	12	132
Kylinga bulbosa P. Beauv.	Bulb Sedge	Cyperaceae	39	1482	36	1260	39	1482	51	2550

Lepidium didymium L.	Lesser Swine-cress	Brassicaceae	17	272	16	240	18	306	0	0
Luzula sylvatica (Huds.)	Great Woodrush	Juncaceae	16	240	7	42	2	2	0	0
Medicago polymorpha L.	Burmedic	Fabaceae	42	1722	75	5550	38	1406	60	3540
Melissa officinalis L.	Lemon Balm	Lamiaceae	0	0	11	110	53	2756	79	6162
Mitracarpus hirtus	Shaggy Buttonweed	Rubiaceae	8	56	10	90	5	20	5	20
Oxalis corniculata L.	Creeping Woodsorrel	Oxalidaceae	0	0	55	2970	41	1640	17	272
Oxalis stricta L.	Yellow Woodsorrel	Oxalidaceae	0	0	5	20	16	240	38	1406
Pavonia urens Cav.	Forest Mallow	Malvaceae	0	0	0	0	0	0	3	6
Physalis peruviana L.	Cape Gooseberry	Solanaceae	0	0	0	0	0	0	4	12
Portulaca oleracea L.	Common Purslane	Portulacaceae	21	420	9	72	12	132	29	812
Salvia hispanica L.	Chia	Lamiaceae	13	156	65	4160	36	1260	17	272
Senna didymobotrya	African Senna	Fabaceae	0	0	16	240	0	0	10	90
Setaria barbata (Lam.) Kunth	bristly foxtail	Poaceae	98	9506	51	2550	66	4290	49	2352
Sida acuta Burm.f.	Common Wireweed	Malvaceae	29	812	13	156	23	506	32	992
Solanum americanum Mill.	American Black Nightshade	Solanaceae	3	6	0	0	0	0	0	0
Tithonia diversifolia (Hemsl.) A. Gray	Mexican Sunflower	Asteraceae	9	72	4	12	11	110	12	132
	TOTAL		N=533	25744	N=811	38284	N=746	35944	N=903	50668
Gini-Simpson index 1- D				**0.91**		**0.94**		**0.9**		**0.94**

d1= distance 1 (0-40m), d2= distance 2 (40-80 m), d3= distance 3 (80-120 m), C= Control site n= species frequency, 1-D= Gini-Simpson diversity index,

4.5.3 Plant species similarity across the investigated distances

Table 4.14 shows the similarity indices observed among the surveyed distances and the control site. The distances of 0 – 40 m and 80 – 120 m downstream from the dump site displayed the highest similarity indices at 90.9%, followed by the distances of 0 – 40 m and 40 - 80 m at 70.0%. On the other hand, the distances of 0 - 40 m and 80 – 120 m recorded a lower similarity index of 69.2%. The distance of 40 - 80 m showed the highest degree of similarity (84.7%) with the control site, while the distance of 0 – 40 m displayed the lowest degree of similarity with the control site.

Moreover, the study identified twenty-nine (29) plant species from the distance of 40 – 80 m, and twenty-six (26) plant species from the distance of 80 – 120 m. Notably, these two distances shared twenty-five (25) plant species. This indicates that the distance of 80 – 120 m contained only one (1) plant species (*Erythrina abyssinica* (Papilionaceae)) not found in the distance of 40 – 80 m. Whereas, the distance of 40 – 80 m had four (4) plant species (*Eucalyptus grandis maidenii* (Myrtaceae), *Euphorbia tirucalli* (Euphorbiaceae), *Furcraea foetida* (Asparagaceae), and *Senna didymobotrya* (Fabaceae)) that were absent in the distance of 80 – 120 m. Moreover, it was observed that distances of 40 - 80 m and 80 - 120 m recorded high percentages of 84.7% and 82.1% respectively, in terms of Sorensen's similarity indices with the control site. A total of twenty-five (25) plant species were found to be common between the distance of 40 – 80 m and the control site, while twenty-three (23) plant species were common between the distance of 80 – 120 m and the control site. In the distances of 40 - 80 m, 80 - 120 m, and the control site, a total of 29, 26, and 30 plant species were recorded, respectively. This indicates that the distance of 40 - 80 m had four plant species (*Euphorbia tirucalli* (Euphorbiaceae), *Hydrocotyle sibthorpioides* (Araliaceae), *Lepidium didymium L* (Brassicaceae), *Luzula sylvatica* (Juncaceae)) not found at the control site. while, the control site contained five plant species (*Erythrina abyssinica* (Papilionaceae), *Guizotia* *abyssinica* (Asteraceae), *Iva xanthiifolia nutt* (Asteraceae), *Pavonia urens cav*. (Malvaceae), *Physalis peruviana L.* (Solanaceae)) that were absent in the distance of 40 - 80 m.

Similarly, the distance of 80 - 120 m included three plant species (*Hydrocotyle sibthorpioides* (Araliaceae), *Lepidium didymium L* (Brassicaceae), *Luzula sylvatica* (Juncaceae)) not observed at the control site. while, the control site contained seven species (*Eucalyptus grandis maidenii* (Myrtaceae), *Furcraea foetida* (Asparagaceae), *Guizotia abyssinica* (Asteraceae), *Iva xanthiifolia nutt* (Asteraceae), *Pavonia urens cav.* (Malvaceae), *Physalis peruviana L*. (Solanaceae), *Senna didymobotrya* (Fabaceae)) that were absent in the distance of 80 - 120 m. With sixteen (16) plant species in common, distance of 0 - 40 m and control site had the lowest Sorensen's similarity index (60.3 %). Thirty species were recorded at the control location, while a total of twenty-three plant species were documented from the distance of 0 - 40 m. This implies that distance of 0 - 40 m had seven (7) plant species (*Beta vulgaris L. (*Amaranthaceae*)*, *Cirsium spinosissimum (*Asteraceae), *Euphorbia tirucalli (*Euphorbiaceae), *Hydrocotyle sibthorpioides (*Araliaceae), *Lepidium didymium L (*Brassicaceae), *Luzula sylvatica (*Juncaceae), *Solanum Amercanum mill.(* Solanaceae)) that were not present at the control site, whereas control site had fourteen (14) species (*Bidens pilosa L. (*Asteraceae), *Brugmansia suaveolens (*Solanaceae), *Commelina diffusa burm.f. (*Commelinaceae*)*, *Digitalia sanguinalis L. (*Poaceae), *Dysphania pumilio (*Amaranthaceae), *Eucalptus grandis maidenii (*Myrtaceae), *Galinsonga quadriradiata (*Asteraceae),*Guizotia abyssinica* (Asteraceae), *Melissa officinalis L. (*Lamiaceae), *Oxalis corniculata L. (*Oxalidaceae), *Oxalis stricta L. (*Oxalidaceae), *Pavonia urens cav. (*Malvaceae), *Physalis peruviana L. (*Solanaceae) that were absent in the distance of 0 - 40 m. These findings indicate that there are no significant variations in plant species composition and distribution across different distances from the dump site, suggesting a minimal impact of leachate from the Cyuve dump site on biodiversity.

Table 4. 15: Sorensen's similarity index of plant species among the investigated distances and control site

Distance (m)	0 – 40 (%)	40- 80 (%)	80 – 120 (%)	Control site (%)
0 - 40	*	70.0	69.2	60.3
40 -80		*	90.9	84.7
80 - 120			*	82.1
Control site				*

CHAPTER FIVE

DISCUSSION

5.1 Introduction

This chapter encompass a comprehensive analysis of the findings obtained from the study. To determine the effects of leachate on soil quality, water quality, and vegetation diversity around the Cyuve dumpsite, Concurrent field visits were conducted simultaneously at the dumping site for collecting the data. The evidence indicated alterations in some physicochemical properties of the soil at the dumping site in comparison to the control site. An analysis of water quality parameters also revealed some variations particularly at the middle stream (MS) station, positioned downstream of the point where leachate entered the stream water. Furthermore, the study showed a notable degree of heterogeneity within plant communities downstream of the dump site.

5.2 Effects of leachate on the soil physicochemical parameters

The findings of this study revealed that there was no significant difference in soil pH among the sampled distances downstream of the dump site, although it was slightly higher within the distance of 0-40 meters closest to the dump site compared to farther distances. Moreover, soil pH values recorded at the sampled distances downstream the dump site were significantly higher than those recorded at the control site, both during the wet and dry seasons. This indicates that the soil pH downstream of the dump site may have been affected by the alkaline leachate observed at the Cyuve waste dump site, as indicated by the results presented in Table 4.1.This is consistent with findings of the study conducted by Omofunmi *et al*. (2020), who noted that old landfills can affect the surrounding soil by producing alkaline leachates with a pH ranging from 8.0 to 8.5. A study conducted by Anikwe and Nwobodo (2002) suggested that the observed increase in soil pH downstream of the dump site could also be attributed to the high organic matter content present in the deposited waste. This organic matter aids in stabilizing the soil by releasing exchangeable cations through organic matter mineralization, thereby helping to mitigate excessive fluctuations in soil pH.

The obtained results are also in agreement with Domínguez et al. (2019) who mentioned that soil pH increases when waste composts are applied to that soil, and noted that this could be because of the presence of carbonates in waste and the organic matter's moderating properties. Furthermore, the present findings are consistent with those of the study carried out by Yahaya et al. (2010), who noted that the rise in soil pH could potentially be attributed to the presence of collapsed cement structures disposed at the site and cement blocks utilized for fencing, acting as a source of calcium carbonate (CaCO3) buffer, along with the influence of rainfall events that dilute the soil solution. The current results align with the results of the study conducted by Nta and Odiong (2017). However, the present results are not in conformity with the result of the study conducted by Emeka *et al*. (2021) who obtained low soil pH in the soil around the dump site and higher soil pH at control site.

Furthermore, the present study revealed a decline in soil pH depth-wise. This decrease could be attributed to the low migration of exchangeable cations and liming material such as Calcium Carbonate (CaCO3) down the soil layers. These results are also in agreement with the study conducted in Akure, Nigeria which showed a decrease in soil pH with depths (Ilemobayo & Kolade, 2008). However, the findings of this study contradict the previous research that was conducted on soil contamination at dumpsites that revealed an increase in soil pH with depth (Taylor, 2014). The higher soil pH observed in the top soil depths compared to the deeper depths may be attributed to the higher content of organic matter (OM) recorded in the top soil samples compared to the deeper soil samples. A study conducted by Fatubarin and Olojugba (2014) highlighted that organic matter acts as a reservoir for exchangeable bases in soil.

The Cation Exchange Capacity (CEC) represents the amount of exchangeable cations per unit weight of dry soil and serves as a crucial determinant of soil fertility. A study by Parameswari *et al.* (2015) highlighted that CEC demonstrates a direct correlation with the soil's capacity to adsorb heavy metals, whereby the adsorption is influenced by both soil properties and the specific characteristics of the element. The present study found that the value of soil CEC within the distance range of 0 to 40 meters was higher compared to values recorded at distances of 40 to 80 meters and 80 to 120 meters in the wet season. Additionally, the CEC values recorded at the sampled distances downstream of the dump site were higher than those recorded at the control site. This indicates that within the distance of 0 to 40 meters nearest to the dump site, there may be a more noticeable decomposition of organic matter, leading to enriched soil nutrients compared to soils found farther away and in the control site. This observation aligns with the results of the study conducted by Yeilagi *et al.* (2021) who reported that the leachate from the dump site enhanced soil quality by elevating fertility indicators such as CEC, total nitrogen (N), available phosphorus (P), potassium (K), and organic matter (OM).

These findings are consistent with those of a study conducted by Anikwe and Nwobodo (2002), which reported that the high cation exchange capacities observed in soils near dump sites could be attributed to the breakdown of waste. This decomposition process of waste tends to generate more exchangeable bases, consequently enhancing soil fertility. Additionally, these results align with the observations of Shiralipour *et al*. (1992), who indicated that high loading rates of solid waste compost can increase soil cation exchange capacity, whereas low loading rates have minimal impact.

In the dry season, the current study observed a decrease in soil cation exchange capacity (CEC) compared to the wet season. Furthermore, there were no significant differences in values of soil CEC across the sampled distances downstream of the dump site compared to the control site. This lack of variation across distances could be attributed to the limited moisture present in the soils, which hampers the breakdown process of organic material. These results align with the study carried out by Fatubarin and Olojugba (2014), which suggested that the low values of CEC during the dry season might result from the reduced distribution of soil organic matter due to low moisture content, as organic matter serves as a reservoir for cations. Additionally, these findings align with the book written Kanehiro and Chang (1956), which noted that dehydration leads to a decrease in soil cation exchange capacity. Moreover, the observed decrease in CEC with depth could be attributed to the reduction in soil organic matter with depth, as suggested by Fatubarin and Olojugba (2014).

The present study revealed that soil organic matter (OM) content was decreased with increasing distance from the dump site. In addition, the higher value of soil OM content was recorded from the distance of 0-40 m that was the closest distance to the dump site compared to the farthest distances and control site. This decline in soil organic matter along the distance gradient may be attributed to the gradual reduction in the quantity of organic materials that were received at each distance from the waste in the dump site. These results aligns with the observations of Shehu-Alimi *et al*. (2020), who reported that higher organic matter content correlates with the breakdown of organic substances. According to the study conducted by Nta and Odiong (2017), the breakdown of organic materials within the dump site might account for the higher levels of organic matter detected in the soil nearest to the dumpsite in contrast to soils located farther away from the dump site. These findings are also consistent with those of the study conducted by Yahaya *et al*. (2010), who noted that the decomposition and composting of various organic materials, including animal waste, vegetable matter, and polymer or plastic materials, may contribute to the increased organic matter content in the soil downstream the dump site in contrast to the control site.

These findings align with the guidelines for educators established by the United States Department of Agriculture in 2014. According to these guidelines, the quantity of OM in soil will increase when the amount of organic materials added exceeds the rate of decomposition. However, if the rate of decomposition surpasses the quantity of organic materials provided, the amount of organic matter in the soil will decrease. The observed decrease in soil OM with depth could be attributed to limited decomposition occurring down the soil profile, possibly due to a lack or shortage of soil microorganisms responsible for decomposition, as suggested by Fatubarin and Olojugba (2014). A study conducted by Parameswari *et al.* (2015) noted that a high presence of OM in the subsoil can hinder the leaching of contaminants, thereby reducing the risks of groundwater contamination through its significant role in adsorption reactions. Whilst, the low levels of soil OM recorded during the dry season compared to wet season may be attributed to reduced moisture content in soil and the burning of waste, which are common occurrences during this period (Fatubarin & Olojugba, 2014). The present results for soil organic matter (OM) in both seasons were higher compared to those reported in the study conducted by Parameswari *et al.* (2015). This difference could be attributed to variations in the composition of waste disposed of in the respective dumpsites.

Electrical conductivity (EC) serves as a measure of salinity and the capacity of a material to conduct charges (Nta & Odiong, 2017). The study's results revealed significant differences among the mean EC values obtained from various sampled distances and depths. Notably, the highest EC mean values were observed at the distance of 0 - 40 m nearest to the dump site, gradually decreasing towards the distance of 80 -120 m. Additionally, higher EC mean values were found at the top depth (0 – 5 cm) compared to deeper depths (15 – 30 cm), indicating that most organic decomposition occurred closest to the dump-site, particularly in the topsoil layer.

A study conducted by Nta and Odiong (2017), reported that the high electrical conductivity in the soil around the dump site may be attributed to an increase in ions and salts from the waste disposed of there. The present study's findings indicate that the observed electrical conductivity has no adverse effect on plant growth around the dump site, as it falls below 0.5 milliScm-1, consistent with the findings of Nta and Odiong (2017). Furthermore, the obtained EC results suggest that the soil in the study area is not saline, as per the salinity classes defined by the USDA Natural Resources Conservation Service, which categorizes soils with EC below 2000 µScm-1 as marginally or non-saline (USDA (United States Department of Agriculture), 2011).

Organic decomposition increases electrical conductivity (EC) in soil by releasing ions such as calcium, magnesium, potassium, sodium, nitrate, phosphate, and chloride as organic matter breaks down. These ions boost the soil’s ion concentration, enhancing its ability to conduct electricity, which EC measures. This indicates that the decomposition of organic materials present in the waste contributed to the observed higher electrical conductivity (EC) levels specifically at the closest distance to Cyuve waste dumpsite, likely due to an increased concentration of ions. However, as noted by Tarchitzky *et al.* (1999), other factors, such as inorganic waste or chemical inputs, can also significantly influence EC, particularly at dumpsites. Thus, the elevated EC near dumping sites is likely a combined result of both organic decomposition and inorganic substances in the waste.

Although several factors affect EC levels in soil, Parameswari *et al*. (2015) indicated that a good soil EC level usually falls between 200 µS/cm and 1200 µS/cm. When the EC level is less than 200 µS/cm, the soil may be sterile and have little microbial activity; when it is more than 1200 µS/cm, there may be a salinity problem because of insufficient drainage. The observed decrease in soil electrical conductivity (EC) in the dry season could be associated to the reduced moisture content in the soil. These results correspond with the study carried out by Ratshiedana *et al.* (2023), which highlighted that soil water content plays an important role in enabling the flow of electrical current within the soil medium. The decrease of electrical conductivity (EC) depth-wise could be attributed to the low leaching of salts to the lower depths. The electrical conductivity (EC) results obtained from the study area during both seasons were higher than those reported in the study conducted by Shehu-Alimi *et al*. (2020) in Nigeria. However, in contrast, these results were lower compared to other studies conducted on municipal solid waste dumpsites by Sam-uroupa and Ogbeibu (2020), Mekonnen *et al*. (2020), and Parameswari *et al. (*2015). The variance could be attributed to differences in the composition of waste deposited in those landfill sites

The present findings revealed a significant difference in total dissolved solids (TDS) among the sampled distances downstream the dumpsite, with the highest values recorded at the distance of 0 – 40 m closest to the dumpsite, and the lowest mean values recorded at the distance of 80 – 120 m. This trend indicates a decrease in soil TDS with distance for both seasons. Additionally, the TDS mean values obtained from the sampled distances were significantly higher than those recorded at the control site. Furthermore, there was a decline in TDS mean values with depth, indicating a downward trend in soil. Thus, the observed decrease in TDS with distance and depth could be ascribed to the increase of soluble ions from the dumped waste being more evident in closer distances and top layers compared to farther distances and deeper depths.

These results align with those of the study conducted by Yeilagi *et al.* (2021), who proposed that the presence of high levels of total dissolved solids in the leachate, including inorganic salts such as calcium, magnesium, potassium, sodium, bicarbonates, chlorides, and sulfates, along with small amounts of organic matter, could explain the notable increase in TDS observed in the dump site closest soil compared to the far away soils, and control site. Additionally, the results align with those of the study conducted Afolagboye *et al.* (2020) who suggested that significant amounts of soluble ions in the waste may contribute to the comparatively high TDS values in the soil around the dump site. The present study recorded lower TDS values compared to the results obtained in the study conducted by Afolagboye *et al*. (2020) in Nigeria. This variation may be due to the differences in the age of the dump sites and the types of waste disposed of in those sites.

The relative proportions of sand, silt, and clay collectively determine a soil's texture (Moorberg & Crouse, 2021). Throughout both seasons, significant differences were observed in the mean percentages of sand, silt, and clay among the investigated distances downstream the dump site. The lowest mean percentages of sand were recorded at the dump site's closest distance (0-40 m), while the highest sand percentages were found at the farthest distance (80-120 m). However, the highest mean percentages of silt and clay were recorded at the dump site's closest distance (0-40 m), with the lowest silt and clay percentages observed at the farthest distance (80-120 m). This indicates that the particle size distribution of the soil downstream from the dump site varied with distance.

Additionally, the present study recorded lower sand percentages and higher silt and clay percentages in the investigated distances at the dumping site compared to the control site. This increased fine content (clay and silt) in the soils near the disposal site could be attributed to the finer particles originating from the higher organic matter content present in these soils compared to the control site. Fine particles resulting from the decomposition of solid waste on the top of the soil may probably contribute to the elevated fine content (clay and silt) in the soil downstream the disposal site (Emeka *et al.*, 2021). These results align with those of the study conducted by Akintola *et al*. (2021), who highlighted that the quantity and quality of organic matter introduced into the soil significantly impact its structure and quality. According to Akintola *et al*. (2021), the soil at the closest dumpsite (0–40 m) would likely possess a more stable structure, higher moisture content, improved water retention and porosity, lower soil strength, and bulk density compared to soils at farther distances (40–80 m, 80–120 m) and control site soils.

Additionally, the results indicated that during both seasons, the mean values of particle size distribution (sand, silt, clay) for soil samples taken at distances of 40 – 80 m, 80 – 120 m, and the control site fell within the sandy loam category on the USDA soil textural triangle (García-gaines & Frankenstein, 2015), or within the silty sand category on Trefethen's Trilinear diagram (Parameswari *et al.*, 2015). Whereas, soil samples collected from the distance of 0 – 40 m were categorized as loam on the USDA soil textural triangle or as clayey silty sand on the Trefethen's soil classification.

5.3 Effects of leachate on the concentration of heavy metals in soil

In the dry season, the present study has recorded higher concentrations of heavy metals (Pb, Cr, Cd) in comparison to the wet season. The present results are consistent with those reported by Yahaya *et al*. (2010), who noted that the dilution of soil solution by rainfall during the wet season, along with the runoff effect, could lead to the removal of heavy metals from the nearby soil of the dump site. Yet, in the dry season, the regular and strong evaporation, coupled with waste burning at the dumping site, may result in a soil solution that is more concentrated in heavy metal ions compared to the rainy season.

Lead (Pb) concentrations at the dumpsite showed the highest mean values within the distance of 0 – 40 meter, while the lowest values were observed within the distance of 80 – 120 meter, during both wet and dry seasons. Furthermore, lead concentration values obtained from distances downstream of the dump site were notably higher compared to those recorded at the control site. Thus, it was observed that lead concentration decreased as distance from the dump site increased. These results align with those of a study by Mekonnen *et al*. (2020), which attributed variations in lead concentrations across distances to differences in soil pH and organic matter content.. The gradual reduction of leachate and lead-containing waste along the distance may also likely contribute to the decrease in lead concentration over those distances. This aligns with the findings of study conducted by Shehu-Alimi *et al*. (2020), which noted that metals exhibit a strong affinity for humic acids, organic clays, and oxides covered in organic materials.

Concentrations of lead (Pb) obtained from the investigated distances downstream the dump site were found to be below 30 mg/kg, the acceptable maximum limit of lead (Pb) concentration in organic fertilizers stipulated by the Rwanda Standards Board (RSB, 2020). Additionally, the obtained results fell within the 2-200 mg/kg range of lead in unpolluted soils as outlined by Nangia and within the 0.3-10 mg/kg range accepted by the World Health Organization (Ediene & Umoetok, 2017). Moreover, the obtained lead concentrations were below the 50-100 mg/kg range indicated by the US EPA standard as the maximum permissible limit for lead concentrations in soil (Mekonnen *et al*., 2020).

The values of lead (Pb) concentrations recorded in the present study were lower compared to other studies conducted by Alain and Naramabuye (2018) in Kigali (Rwanda), Mekonnen *et al*. (2020) in Tepi town (Ethiopia), and Sam-uroupa and Ogbeibu (2020) in Benin Metropolis (Nigeria), which reported concentrations of 443.4 ppm (mg/kg), 3.26-57.56 (mg/kg), and 2.09-2.3 mg/kg, respectively. These variations could be explained by differences in the quantity and composition of wastes containing lead, such as plastics, electronic waste, batteries, paints containing lead, and pipes, which are dumped in landfills without being separated. This aligns with the study conducted by Mehmet and Acarer (2022), who noted that leachate composition is significantly influenced by various factors such as category, age, humidity, height, dumping technique, and weather.

The present findings revealed that the highest values of chromium were obtained at a distance of 0 – 40 meters closest to the dump site, while the lowest values were recorded from the distance of 80 – 120 meters for the wet and dry seasons. This indicates that that the concentration of chromium was higher in the soil nearer to the dump site compared to the soil at greater distances. This increase in chromium concentrations at the closest distance to the dump site could be assigned to the volume of leachate having a high concentration of chromium ions from the dumped waste that was much leached into the underlying soils of the closest distance compared to the farthest distance as suggested by Umoh and Etim (2013). The results are also in conformity with the research carried out by Awokunmi *et al.* (2010) which disclosed the reduction in the level of chromium along distances downstream the dump site.

Furthermore, the current study indicated a significantly greater concentrations of chromium in the soil downstream the dump site than the chromium level measured at the control site, which indicates a pollution of the soils around the dump site by the ions of chromium from the waste disposed in the dump site. The present results are also in conformity with those of with the research carried out by Awokunmi *et al.* (2010) who reported that the increased level of chromium in the soil closest to the dump site could be attributed to the improper disposal of automobiles and electronic wastes in the dump site, which facilitated the leaching of chromium ions particularly in the soils nearest to the dump site.

The chromium concentrations obtained in the investigated distances downstream the dump site were found to be below 50 mg/kg, the acceptable maximum limit of lead (Pb) concentration in organic fertilizers stipulated by the Rwanda Standards Board (RSB, 2020), and below 400 mg/kg, the USA standard for uncontaminated soils (Ediene & Umoetok, 2017). However, they were found to be higher than the 0.002 - 0.2 mg/kg acceptable limits prescribed by the World Health Organization (Ediene & Umoetok, 2017). These results were further found to be within the 5-3000 mg/kg unpolluted range as noted by Nangia (Ediene & Umoetok, 2017).

The obtained chromium concentration results were lower compared to the results obtained in other studies conducted on municipal solid waste dumpsites by Alamri (2023), Alain and Naramabuye (2018), and Awokunmi *et al.* (2010). However, the obtained results were higher than the results obtained in the study conducted in Nigeria by Sam-uroupa and Ogbeibu (2020). These differences might be associated with the age of the dump sites and the existence of chromium-containing wastes, including stainless steel, protective coatings on metal, magnetic tapes, and pigments used in various materials such as paints, cement, paper, rubber, composition floor covering, and others, which were commonly dumped in these landfills (Mehmet & Acarer, 2022).

Although cadmium is an uncommon heavy metal, it is considered one of the most harmful to human health (Agbeshie *et al.*, 2020). The present study findings revealed a decline in cadmium concentrations along the investigated distances downstream from the dump site, with the highest concentrations recorded at a distance of 0 – 40 m and the lowest concentrations at a distance of 80 – 120 m for both wet and dry seasons. Additionally, the measured concentrations of cadmium downstream from the Cyuve dump site were greater than those observed at the control site. This suggests that the leachate from the Cyuve dump site is gradually contributing to the elevation of cadmium level in the soil nearer to the Cyuve dump site. These results are in agreement with those of the conducted study by Umoh and Etim (2013), who noted that the variation in heavy metal concentrations along the distance could be explained by differences in the quantity of leachate and metal-containing wastes received at each of the investigated distances, which eventually leach and elevate the concentration of those heavy metals in the underlying soils.

Concentrations of cadmium obtained in the present study were below 5 mg/kg, the acceptable limit of cadmium in organic fertilizers prescribed by the Rwanda Standards Board (RSB, 2020). Additionally, except for the samples from the distance of 0-40 m, the results fell within the 0.002-0.5 mg/kg range prescribed by the World Health Organization (WHO) for unpolluted soils, and within the uncontaminated range of 0.01-0.7 mg/kg outlined by Nangia (Ediene & Umoetok, 2017). Furthermore, all cadmium concentration results obtained at the investigated distances were below 1.4 mg/kg, which is the permissible limit prescribed by the United States Environmental Protection Agency (Mekonnen *et al*., 2020).

The present study revealed low concentrations of cadmium in the soil downstream the Cyuve dump site compared to the other tested heavy metals. This could be due to the reduced concentration of cadmium in the leachate. The observed lower cadmium concentration aligns with the results of the study conducted by Agbeshie *et al.* (2020) who reported that low concentration of cadmium in the soil around the dump site could be linked to the low cadmium materials deposited from motor lubricants, sludge, batteries, PVC materials, and coatings.

Moreover, the concentrations of cadmium obtained from the distance of 0 – 40 m were slightly higher compared to the study conducted by Sam-uroupa & Ogbeibu (2020) in Benin Metropolis (Nigeria). However, these results were lower compared to other studies conducted on municipal solid waste dumpsites by Alain & Naramabuye (2018) in Kigali (Rwanda), and Mekonnen *et al*.

(2020) in Tepi town (Ethiopia), which reported concentrations of 20.96 mg/kg, and 0.53-2.26 mg/kg, respectively. The differences could be related to disparities in the ages of the disposal sites and the dumping systems of wastes containing cadmium, such as paints, batteries, and plastics, as indicated in the studies conducted by Samadder *et al.* (2017), and Mekonnen *et al.* (2020). Therefore, the present investigation revealed that as the distance from the pollution source increased, the levels of contaminants in the soil downstream from the Cyuve landfill declined.

The neutral to slightly alkaline pH recorded at various depths of the investigated distances downstream the dump site, the exchangeable acidity of the soils might be low. The results of this study indicate a slight increase in heavy metals depth-wise, which could be associated with the observed decrease in soil pH with depth. This decrease in soil pH may result in reduced adsorption of heavy metals in the soil at lower depths. These findings are consistent with those of Taylor (2014), who noted that decreasing soil pH reduces the adsorption of heavy metals, thereby increasing their concentration and distribution in the soil solution.

The present findings also align with those of the study conducted by Naveen *et al*. (2018), who indicated that heavy metal levels are typically higher at low pH levels. A research carried out by Amano *et al.* (2021) also noted that heavy metals are typically more soluble at lower pH levels, rendering them more hazardous. Low soil pH typically facilitates the distribution and movement of metals within the soil, whereas elevated soil pH can restrict the mobility of certain types of metals throughout the soil (Ilemobayo & Kolade, 2008). Generally, soil pH significantly influences the bioavailability, toxicity, and leaching capacity of heavy metals into nearby soil (Emeka *et al.*, 2021).

Furthermore, a study conducted by Alain and Naramabuye (2018) noted that when it rains on sandy loam soils, heavy metals are redistributed within the soil horizons and may penetrate deeper into the soil, potentially contaminating groundwater. This indicates that soil texture may be a contributing factor. The observed decrease of lead (Pb) and chromium (Cr) depth-wise is consistent with the results of Louhar *et al.* (2019), who noted that the type of soil and properties of the elements determine the distribution of heavy metals at different depths. Additionally, a study by Shehu-Alimi *et al*. (2020) indicated that factors such as pH, metal quantity, soil CEC, organic carbon content, oxidation state of mineral components, and the system's redox potential influence the solubility of metals in soils.

5.4 Effects of leachate on the water quality parameters

The present study revealed that the leachate from Cyuve dumpsite was alkaline, as indicated by the high pH values shown in Table 4.10. This suggests that Cyuve dumpsite is old, having been in operation for more than five years. As per the research conducted by Amano *et al.* (2021), a landfill site is considered young if its leachate pH is less than 6.5, meaning it is less than five years old. On the other hand, if the leachate pH is above 7.5, it is considered old and the increase in pH is attributed to methane generation. In addition, a study conducted by Parvin and Tareq (2021) also noted that pH of the leachate can also vary based on the age of the landfill and the quantity of volatile acid present because of methanogenic bacteria. For instance, a study conducted by Parvin and Tareq (2021) noted leachate from young dump sites varies in pH from 5.0 to 6.5, while leachate from mature dump sites ranges in pH from 7.8 to 8.64.

The present results indicate that the water pH obtained from the MS and DS stations was slightly higher compared to the US. This increase in stream water pH at these stations (MS and DS) could be attributed to the introduction of leachate. These findings align with those of the study carried out by Omofunmi *et al*. (2020), which noted that the composition and volume of generated leachates, as well as the proximity of the dumpsite to water bodies such as groundwater and surface water, influence the extent of contamination in such water bodies. A study carried out by Bangani *et al*. (2023) also noted that the waste characteristics of a specific dump site can impact the pH of the water body into which its leachate enters. Emmanuel-akerele and Peter (2020) emphasized the importance of water pH, as variations in pH levels can affect the toxicity of microbial toxins present in the water. Additionally, a study conducted by Mekonnen *et al.* (2020) indicated that a higher pH range indicates greater water productivity.

The present study findings indicated that the water pH values observed at the sampled stations fall within the (5.5 – 9.5) range set by the Rwanda Standard Board for natural potable water (MININFRA, 2019a). Furthermore, they align with the (6.5 – 8.5) and (6-8.5) ranges recommended by the WHO and US EPA standards, respectively, for predominantly fresh waters (Emmanuel-Akerele & Peter, 2020; Theodoros & Juergen, 2018). The pH values obtained for both seasons were greater than those noted in a study carried out in Lagos by Emmanuel-Akerele and Peter (2020), but they were consistent with the findings of another study conducted in Tepi Town by Mekonnen *et al*. (2020) concerning the influence of a solid waste landfill on the nearby soil and the quality of river water.

Electrical conductivity in water is a measure of its ability to conduct an electrical current (Kate, 2019). The elevated electrical conductivity values observed in the leachate during both seasons may be attributed to the significant concentration of dissolved inorganic compounds in their ionized form within the disposal site. According to a study by Mekonnen *et al.* (2020), high electrical conductivity values in leachate can indicate the presence of contaminants from ionizable materials such as potassium, alkalis, chlorides, sulfides, carbonate compounds, and sulfate in the waste disposed of at the dumpsite.

The electrical conductivity results obtained from the MS (middle stream) and DS (downstream) stations were notably higher than those from the US (upper stream). This indicates the entrance of leachate probably containing a substantial quantity of ionizable substances from the waste within the dump site. Additionally, in 2003, the U.S. Environmental Protection Agency (EPA) stated that an increase in water electrical conductivity is caused by an increase in ions such as carbonate, bicarbonate, chloride, sulfate, nitrate, salt, potassium, calcium, and magnesium. The electrical conductivity results from this study were higher to those of a study conducted by Emmanuel-Akerele and Peter (2020) in Lagos, and the study conducted in Tepi Town by Mekonnen et al. (2020). However, these results remained below the maximum permissible limits of water electrical conductivity set by the Rwanda Standard Board and WHO standards, which are 2500 µS/cm and 1000 µS/cm respectively, for natural potable water (MININFRA, 2019a; Emmanuel-Akerele & Peter, 2020).

Total dissolved solids (TDS) is a measure of the amount of inorganic salts and very small quantities of organic materials dissolved in water; and it shows low impurity levels (Emmanuel-akerele & Peter, 2020). The findings indicated that during both wet and dry seasons, the upper stream (US) station that was considered as control recorded lower TDS values than the middle stream (MS) and downstream (DS) stations as presented in Table 4.10. This could have resulted from the entrance of the leachate containing the inorganic salts and organic matter from the dumpsite into the nearby stream through this MS station. The results align with those of the investigation conducted by Adeolu *et al*. (2011) who noted that in addition to making water less palatable, high

TDS concentrations can irritate human gastrointestinal tracts and have laxative effects, especially during transit.

However, the obtained results of TDS along the river were below the 1500 mg/l and 1000 mg/l maximum permissible limits prescribed by the Rwandan standards Board and WHO standard respectively for natural potable water (Emmanuel-akerele & Peter, 2020; MININFRA, 2019a; Mosley *et al*., 2005). Thus, there are moderate levels of dissolved salts in the water of the sampled stream. The present results of TDS are lower than the results obtained in the study conducted in Tepi town by Mekonnen *et al*. (2020). On the other hand, these results were higher than the TDS mean value obtained in the study conducted by Emmanuel-akerele & Peter (2020) on the microbial and physicochemical evaluation of soil and water in proximity to waste disposal sites in Lagos.

Dissolved oxygen relates to oxygen molecules that have been absorbed in water (Mesner & Geiger, 2005). The results of the study indicated lower values of dissolved oxygen (DO) in leachate samples during both seasons, as presented in Table 4.1. Additionally, the study found that the upper stream (US) station recorded higher dissolved oxygen values than the middle stream (MS) and downstream (DS) stations in both seasons. This indicates an increase in oxidizable organic matter entering the stream from the waste within the dump site, consequently decreasing the level of dissolved oxygen in the water at the MS and DS stations.

These findings align with those of the study conducted by Bangani *et al*. (2023), who observed that waste reduces the amount of dissolved oxygen (DO) in the water body it enters. It is believed that one of the primary effects of leachate discharge into water bodies is oxygen depletion in surface water, which can impact the fauna and flora of the stream bottom and lead to ammonia toxicity (Parvin & Tareq, 2021). A study conducted by Perwira *et al*. (2020) also noted that dissolved oxygen decline in water when organic matter is converted into ammonia and nitrate.

Furthermore, these findings are consistent with the Environmental Protection Agency of Ethiopia (EEPA, 2003), which indicate that the amount of suspended material in the water can affect the saturation concentration of dissolved oxygen both physically, by reducing the available water volume for solution, and chemically, due to the ability of suspended particles to obtain oxygen.

It was noted in the study conducted by Olarewaju *et al.* (2012) that a concentration of dissolved oxygen (DO) less than 5 mg/l can have adverse effects on aquatic biological life, while a concentration below 2 mg/l may result in the death of most fish. Additionally, as noted by Galal-Gorchev (1993), a decrease in dissolved oxygen levels within a body of water may facilitate microbial reduction of nitrate to nitrite and sulfate to sulfide, consequently increasing the concentration of ferrous iron in the water. The dissolved oxygen (DO) results obtained along the Cyuve dump site nearby stream water were above 5 mg/l, which is the minimum level of dissolved oxygen required according to the standards set by the Rwandan Standard Board for natural potable water (MININFRA, 2019b).

Chemical oxygen demand indicates the quantity of oxygen needed to oxidize organic materials present in water (LaDuke, 2019). The present study revealed that, the upper stream (US) station consistently recorded lower chemical oxygen demand (COD) values compared to the middle stream (MS) and downstream (DS) stations throughout both wet and dry seasons. This indicates that the influx of leachate containing chemicals and oxidizable organic matter from the waste in the dump site into the stream is responsible for the elevated COD levels at these stations. Consequently, large quantities of oxygen are required to convert all of these organic molecules into CO2 and water, thereby increasing COD concentrations.

These results align with those of the study conducted by Perwira *et al.* (2020), which observed that ammonia, nitrate, and COD concentrations rose alongside an increase in total organic matter in the water, while dissolved oxygen levels decreased. This phenomenon may be attributed to oxygen consumption during the degradation of both organic and inorganic compounds in the water. Additionally, the present results align with those of the study conducted by Maqbool *et al.* (2011), who suggested that the degree of contamination in water bodies can be more accurately assessed by measuring modified chemical oxygen demand (COD). The obtained results also correspond with a study conducted in Tepi town, Ethiopia by Mekonnen *et al*. (2020). Furthermore, the COD results obtained along the stream exceeded the maximum limits for COD in potable water prescribed by the World Health Organization (WHO) (2004) standard and the Rwandan Standard Board (RSB), which are 15 mg/l and 5 mg/l, respectively (Mekonnen *et al*., 2020; (MININFRA, 2019b).

The present study's findings revealed that the upper stream (US) station recorded lower number of total plate count, total coliforms, and *Escherichia coli* bacteria during both wet and dry seasons compared to the middle stream (MS) and downstream (DS) stations. The higher densities of bacteria found in the leachate from the Cyuve dumpsite, as shown in Table 4.1, likely contribute to the high bacterial concentrations observed at the MS and DS stations along the stream.

Therefore, it is probable that the middle stream station (MS) is receiving significant bacterial loads due to the inflow of leachate from the landfill. The discharge of leachate into the water stream may result from improper waste disposal practices, leading to the transfer of sewage and wastewater from the dump site into the nearby stream (Kusari, 2019; Parvin & Tareq, 2021). These findings align with the results of the study conducted by Emmanuel-akerele and Peter (2020), which reported that discharges from waste materials in dump sites could be a contributing factor to the higher bacterial loads found in neighboring rivers. Additionally, as stated by Emmanuel-Akerele and Peter (2020), effluents from septic tanks and inappropriate disposal of sewage and wastewater originating from household activities could also be factors contributing to the elevated abundance of these bacteria in water sources.

Furthermore, the results of total viable count, total coliform, and E. coli obtained from the middle stream (MS) station were significantly higher than those obtained at the downstream station (DS). This finding aligns with the results reported by Vodyanitskii (2016), which noted a decrease in microbial community structure with increasing distance from the pollution source. Moreover, present study revealed that bacterial concentration results exceeded the maximum allowable limits in potable water established by the World Health Organization (WHO) and the Rwandan Standard Board (RSB), rendering the stream unsuitable for human consumption due to high bacterial loads.

According to recommendations from the RSB and WHO, potable water should have no *Escherichia coli* or coliform bacteria, with a heterotrophic bacterial count (total viable count) of less than 50 CFU/mL or 20 CFU/mL, respectively (Curutiu *et al.*, 2019; MININFRA, 2019a). Furthermore, since coliforms are considered primary indicators of water pollution, the US Environmental Protection Agency (EPA) recommends that water bodies in which they are found should not be used for drinking purposes (Emmanuel-Akerele & Peter, 2020).

Most importantly, Emmanuel-Akerele and Peter (2020) highlighted several significant human pathogens, such as gastroenteritis, typhoid fever, dysentery, and urinary tract infections, that are associated with the presence of fecal coliforms, including *Escherichia coli*, in water. It should be noted that a persistently positive test result for *Escherichia coli* or fecal coliform organisms in water indicates an immediate need for remediation (Mosley *et al*., 2005).

5.5 Effects of leachate on heavy metal concentrations in the stream water

The findings of the present study revealed high levels of lead and chromium in the leachate compared the obtained levelsof these heavy metals in stream water in both wet and dry seasons. Additionally, the present study results showed that along the dump site nearby stream, the upper stream (US) consistently recorded lower lead and chromium concentrations compared to the middle stream (MS) and downstream (DS) stations throughout both seasons. The increase in lead and chromium concentrations in stream water at the MS and DS stations compared to the upper stream (US), might be attributed to the entry of leachate containing heavy metal ions from the waste disposal site, primarily at the MS station. These results align with those of the study conducted by Mekonnen *et al.* (2020), which noted that the rise in these heavy metals could be linked to the quantity and composition of waste materials, including electronic waste, lead batteries, lead-containing paints, pipes, and plastics, that are carelessly dumped in the dump site and subsequently find their way into the nearby watercourse.

Furthermore, Kusari (2019) noted that leachate contains contaminants that are often present in high concentrations and can adversely affect water quality. Additionally, a study conducted by Mehmet and Acarer (2022) also noted that the duration of operation of the dumpsite, the amount of waste it contains, and the types of waste, such as electronic waste, batteries, paints, stainless steel, protective coatings, magnetic tapes, paint pigments, cement, paper, rubber pipes, and plastics, improperly disposed of there, might contribute to the rise in heavy metal levels within the nearby watercourse.

In comparison to the wet season, the dry season's water samples showed higher quantities of lead and chromium. According to the study conducted by Olafisoye *et al.* (2013), this could be attributed to the slow water currents during the dry season, which allow heavy metals to settle and accumulate in water without turbulence. The concentrations of lead found in the leachate and stream water were higher than those obtained in Mekonnen *et al*.'s study conducted in Tepi Town in 2020. The lead concentration results at the MS and DS stations exceeded the maximum allowable limits for natural potable water set by the Rwandan Standards Board and the World Health Organization at 0.01 mg/l and 0.05 mg/l, respectively (Mekonnen *et al*., 2020; MININFRA, 2019a). Similarly, the chromium values exceeded the maximum allowable limit of 0.05 mg/l set by the Rwandan Standard Board (MININFRA, 2019a).

However, levels of lead (Pb) and chromium (Cr) were revealed to be within the accepted standards of 8.5 mg/l and 50 mg/l, respectively, set by the United States Environmental Protection Agency for Class III surface water intended for fish consumption, recreation, propagation, and maintenance of a healthy population of fish and wildlife (Nsabimana *et al.*, 2020; Theodoros & Juergen, 2018). While lead and cadmium have no biological function and are highly harmful to both humans and aquatic organisms, chromium is essential for the metabolism of all living organisms, including humans, although it becomes toxic at excessive levels (Nsabimana *et al*., 2020).

The present study that cadmium concentration was below the detection level at all sampled stations (US, MS, DS) along the stream. This indicates that the stream water had cadmium concentrations below the detection limit, indicating that the presence of this metal in the stream is not a significant concern as it remains below the maximum allowable limit of 0.003 mg/l for natural potable water set by the Rwandan Standard Board (MININFRA, 2019a). The low cadmium concentration detected in both the leachate and the nearby stream may be attributed to the limited volume of waste materials containing cadmium, such as paint, batteries, and plastics, disposed of in the Cyuve dump site. Cadmium, as highlighted by Nsabimana *et al.* (2020), serves no biological function and is extremely harmful to both humans and aquatic life. Its high concentration in water can also negatively impact egg production, leading to a decrease in offspring and ultimately causing a decline in population and productivity.

5.6 Effects of leachate on the vegetation diversity

The results of this study suggest that the Simpson diversity indices obtained from the plant communities downstream of the Cyuve dump site varied between 0.90 and 0.94. This indicates that the plant species within these communities show a high degree of heterogeneity, as they fall within the range of 0.81 to 0.99 as outlined in the guidelines for Interpreting Simpson Diversity Index Scores (Guajardo, 2015). Despite changes in soil physicochemical properties due to leachate from the landfill, plant species diversity at the dump site was similar to that of the control site, suggesting that the leachate did not significantly reduce plant diversity. This high diversity of plant species could be attributed to the favorable climate of Rwanda's Northern Province, which has abundant rainfall, promoting species growth (Twahirwa et al., 2023). Such findings are consistent with Ariyo's (2020) observation that high rainfall correlates with greater plant species diversity in wet locations.

Additionally, propagules (such as seeds and spores) play a critical role in colonizing and regenerating disturbed sites like dumpsites, which helps explain the lack of significant differences in the presence of annual herbs between the dump site and the control site. Propagule dispersal mechanisms, such as wind, water, or animals, often allow seeds and spores to spread across varying distances, promoting regeneration (Hopfensperger, 2007). The ability of certain plants, including annual herbs, to disperse seeds widely allows them to thrive in both the dumpsite and control site, even when soil conditions differ. Omoigui and Onyeibor (2019) observed that high organic matter in disturbed soils, such as those at dumpsites, can also enhance plant growth, supporting propagule establishment.

Moreover, the comparable existence of annual herbs at the Cyuve dumpsite and control site may be influenced by the resilience of herbaceous plants and their propagules, which can withstand variations in soil pH or nutrient availability (Hopfensperger, 2007). Studies indicate that even in degraded soils, the regeneration capacity of annual herbs can remain robust if propagule banks (seeds, spores) are abundant and the plants are well-adapted to local conditions (Bischoff & Scholten, 2020). This analysis highlights how the presence of propagules such as seeds, spores, or other reproductive material coupled with frequent disturbances in the environment, creates favorable conditions for the proliferation of various herb species. Therefore, findings highlight the importance of propagules in maintaining species diversity, even in areas affected by waste leachate, where conditions are less favorable than natural sites.

Furthermore, Sorensen's similarity indices above 50% between the dumpsite and control site underscore the strong similarity in plant species composition between the two areas (Messou et al., 2012), despite the environmental disturbances. This points to the significant role of propagule dispersal in maintaining plant diversity and ensuring the regeneration of annual herbs and other species across both disturbed and undisturbed areas (Ariyo, 2020).

5.7 Limitation of the study

The present study was conducted over a limited timeframe to align with the time boundaries set by the university for master's students conducting research, this may not fully capture the long-term effects of waste dumping on soil, water quality and plant diversity. Longer-term monitoring could reveal trends and dynamics that might not be evident in a shorter study period.

CHAPTER SIX

CONCLUSIONS AND RECOMMENDATIONS

6.1 Conclusion

The study's findings highlighted the significant influence of leachate from the Cyuve open dump site on both soil and water quality, as well as on plant diversity in the study area. Leachate has notably affected soil physicochemical parameters such as pH, EC, TDS, and CEC, with higher levels of these parameters observed downstream from the dump site compared to control site. Moreover, heavy metal concentrations, including Pb, Cr, and Cd, were relatively higher downstream from the dump site than the control site, indicating potential leachate influence, though they were not exceeding established standards.

Leachate samples showed higher concentrations of heavy metals, particularly lead and chromium, posing a serious threat to nearby water streams, especially at middle and downstream locations. Furthermore, the study noted that plant species diversity of plant communities downstream the dumpsite were not significantly affected, despite the observed effect of leachate on soil physicochemical properties. This suggests that the levels of heavy metals and chemical contaminants, have likely not reached levels to reduce the nutrient availability or disrupt the soil structure. In conclusion, presence of contaminants, including heavy metals, in the leachate from the Cyuve dump site is affecting the nearby soil and water physicochemical characteristics, microbial indicators, and ecosystem health.

6.2 Recommendations

Based on the findings of the study, several recommendations can be made to address the environmental challenges posed by the Cyuve open dump site and mitigate its impacts:

i. **Implement phytoremediation technologies**: Considering the soil is becoming contaminated with heavy metals and other pollutants from the Cyuve dump site, the adoption of phytoremediation techniques should be considered. This entails employing particular plants recognized for their capacity to uptake pollutants from the soil. Planting such species especially those indigenous to the area in the affected areas can help in the remediation process and restore soil health over time.

ii. **Enforce regulations against indiscriminate waste discharge**: Enforcing strict regulations against illegal waste disposal is essential to prevent environmental damage and landfill disasters. This involves penalizing offenders and implementing monitoring systems to detect and prevent illegal dumping.

iii. **Promote waste reduction, recycling, and reuse**: Public awareness campaigns and educational programs should be initiated to promote waste reduction practices, encourage recycling efforts, and emphasize the importance of reusing materials. Community participation and engagement are crucial in fostering a culture of responsible waste management.

iv. **Establish sanitary or engineered landfill sites:** There is a pressing requirement for the establishment of proper waste management facilities, such as engineered landfill sites. These facilities should adhere to strict environmental standards and be equipped to prevent leachate pollution of soil and water reserves.

v. **Strengthened Legal and Regulatory Framework:** To prevent disasters like the Kiteezi landfill incident, proper landfill management must prioritize effective leachate control, fire hazard prevention, and regular maintenance. Implementing waste segregation, gas collection systems, and disaster preparedness plans will reduce environmental and safety risks. Strengthening regulations, ensuring compliance, and promoting alternative waste management strategies can further minimize reliance on overburdened landfills.

vi. **Collaborate with stakeholders**: Collaboration among government agencies, local communities, environmental organizations, and other stakeholders is essential in addressing the challenges posed by open dump sites effectively. This includes coordinating efforts to develop and implement comprehensive waste management plans, allocate resources for infrastructure development, and monitor the effectiveness of remediation measures.

vii. **Invest in research and innovation**: Continued research and innovation in waste management technologies and practices are essential for finding sustainable solutions to environmental pollution. Investing in research initiatives focused on alternative waste treatment methods, cleaner production techniques, and innovative remediation technologies can assist in tackling the underlying reasons of environmental degradation and pave the way for a more sustainable future.

viii. **Further Research Studies:** It is recommended that future studies focus on the long-term impact of leachate from the Cyuve dump site on groundwater quality and aquatic ecosystems. Comprehensive data on leachate's long-term impacts can guide policymakers in waste management and raise public awareness to foster community engagement in sustainable practices.

By implementing these recommendations, stakeholders can work towards mitigating the adverse effects of open dump sites on soil and water quality, protecting public health, and promoting environmental sustainability in the study area and beyond.

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APPENDICES

Appendix 1: Some field pictures that were taken during the data collection



Titration in the lab, Quadrat for plant species sampling, Leachate sampling with the sampling cooler box respectively



Usage of soil Auger for soil sampling in wet and dry seasons, and soil sample preparation respectively at the laboratory

Cooler box for water and leachate samples transportation, quadrat sampling method used respectively

Water sampling

Appendix 2: A letter from the laboratory

UNIVERSITY of RWANDA

COLLEGE OF SCIENCE AND TECHNOLOGY

SCHOOL OF SCIENCE

OFFICE OF THE DEAN

TO WHOM IT MAY CONCERN

RE: RESEARCH PROJECT LABORATORY WORK

Dear Sir/Madam,

This is to confirm that Mr. TUYIZERE Naphtar, a student at Kyambogo University with registration number 21/X/GMSM/14661/PE pursuing a Master of Science in Conservation and Natural Resources Management did his laboratory work of water quality analysis by using Biology and Chemistry laboratories at the University of Rwanda, College of Science and Technology, School of Science.

for Associate Professor Denis NDANGUZA

Dean, School of Science, UR-CST.

UNIVERSITY OF RWANDA School Of Sciences COLLEGE OF SCIENCE AND TECHNOLOGY

Date: 11../07/2023

Appendix 3: Research introductory letter

KYAMBOGO UNIVERSITY

P. O. BOX 1 KYAMBOGO
Tel: 041 - 4286792 Fax: 256-41-220464
Website :www.kyu.ac.ug Email: drgt@kyu.ac.ug

Directorate of Research and Graduate Training

Office of the Director

APPENDIX 8

Date: 24/03/2023

TO WHOM IT MAY CONCERN

RE: TUYIZERE NAPHTAR

Dear Sir/Madam,

This is to introduce to you the above named student Reg: No 21/X/GMSM/14661/PE pursuing Master of science in Conservation and Natural resource management. Department of BIOLOGICAL SCIENCE, Kyambogo University.

She/he intends to carry out research on EFFECTS OF WASTE DUMPING ON ENVIRONMENTAL COMPARTMENTS in partial fulfillment of the requirements of the award of MASTER OF SCIENCE IN CONSERVATION AND NATURAL RESOURCES MANAGEMENT.

The purpose of this letter therefore is to request you to grant him/her permission to carry out his/her study in your institution.

Any assistance rendered to her/him will be highly appreciated.

Yours sincerely,

Prof. Bosco Bua
AG. DIRECTOR

KYAMBOGO UNIVERSITY
24 MAR 2023
DIRECTOR
DIRECTORATE OF RESEARCH AND GRADUATE TRAINING