SPATIAL ANALYSIS AND QUANTIFICATION OF MICROPLASTICS IN THE NAIROBI RIVERS AND WASTEWATER TREATMENT PLANTS

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Spatial Analysis and Quantification of Microplastics in the Nairobi Rivers and Wastewater Treatment Plants

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DECLARATION

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

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This thesis has been submitted for examination with our approval as the University Supervisors.

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DEDICATION

This work is dedicated to my mother, Linah Mengich, who has always believed in me and taught me the importance of perseverance in life. In loving memory of my father, Paul Kigen, may you continue to rest peacefully, dad.

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ABBREVIATIONS AND ACRONYMS

ANOVA	Analysis of variance
ATR-FTIR	Attenuated Total Reflectance Fourier Transform Infrared
FAO	Food and Agriculture Organisation
HDPE	High Density Polyethylene
HRT	Hydraulic Retention Time
JKUAT	Jomo Kenyatta University of Agriculture and Technology
LDPE	Low Density Polyethylene
MP	Microplastic
FPA-FTIR	Focal Plane Array Fourier Transform Infrared
FTIR	Fourier Transform Infrared Spectroscopy
NACOSTI	National Commission for Science, Technology, and Innovation
NCWSC	Nairobi City Water and Sewerage Company
NEMA	National Environment Management Authority
NOAA	National Oceanic and Atmospheric Administration
PBDEs	Polybrominated Diphenyl Ethers
PCBs	Poly-Chlorinated Biphenyls
PE	Polyethylene
PES	Polyester
PET	Polyethylene Terephthalate
pH	Potential of Hydrogen
POPs	Persistent Organic Pollutants
PP	Polypropylene
PS	Polystyrene
PUR	Polyurethane
PVC	Polyvinylchloride
Pyr-GC-MS	Pyrolysis-Gas Chromatography-Mass Spectrometry
QGIS	Quantum Geographic Information System
SEM	Scanning Electron Microscope
SEM-EDS	Scanning Electron Microscope Energy-Dispersive Spectroscopy
UK	United Kingdom

USA	United States of America
UNEP	United Nations Environmental Programme
WHO	World Health Organization
WPO	Wet Peroxide Oxidation
WWTPs	Wastewater Treatment Plants

ABSTRACT

Microplastics have raised significant concerns due to their potential threats to human health and aquatic species. The accumulation of microplastics in aquatic environments has become increasingly alarming in recent years. While microplastics have been widely detected in various water bodies over the past few decades, studies in many developing countries, including those in Africa, remain limited. This study aimed to determine the abundance and composition of microplastic particles in selected rivers and in two wastewater treatment plants in Nairobi, Kenya. Water samples were collected from both the rivers and the wastewater treatment plants, using grab sampling technique. The microplastic particles, ranging in size from 4.75 mm to 300 µm, were categorised through visual identification with a dissecting microscope and characterised using Fourier Transform Infrared Spectroscopy (FTIR). Pearson Product-Moment Correlation analysis was then employed to investigate the relationship between microplastic concentration and land use. The abundance of microplastics in the rivers ranged from 0.22±0.026 to 0.917±0.061 particles/litre. Fibres (51.15%) and white particles (40.08%) were the most common types of microplastics detected, with polyethylene (PE) and polystyrene (PS) being the predominant polymers. Higher concentrations were noted near industrial, commercial, and residential areas, particularly in informal settlements along the rivers. A positive correlation was observed between microplastic abundance and built-up areas, while a negative correlation was found with forested areas and wetlands. In the studied wastewater treatment plants, the average abundance of microplastics in the influent at Ruai was 9.99 ± 0.50 particles per litre, and 4.7 ± 0.28 particles per litre in Kariobangi. This abundance significantly decreased in the final effluent to 1.08 ± 0.18 particles/litre in Ruai and 1.03 ± 0.08 particles/litre in Kariobangi, corresponding to removal efficiencies of 89 ± 1.91% and 78 ± 3.16%, respectively. Despite these reductions, the wastewater treatment plants were releasing approximately 123.1 ± 20 million and 10.3 ± 0.91 million microplastics per day into the receiving river. These findings highlight the influence of anthropogenic activities and wastewater treatment plants on the abundance of microplastics in the Nairobi River Basin. This study represents one of the first detailed assessments of microplastic distribution in Nairobi’s freshwater systems and provides critical baseline data on microplastic pollution in urban rivers of Sub-Saharan Africa. It underscores the urgent need for improved wastewater management and pollution control strategies, as the presence of microplastics poses a serious ecological risk and potential implications for human health.

CHAPTER ONE

INTRODUCTION

1.1 Background Information

Plastic pollution is a significant environmental challenge that impacts both terrestrial and aquatic ecosystems globally. Since the mid-20th century, the production and use of plastic have increased tremendously. According to Plastics Europe (2024), a major pan-European plastic association, worldwide plastic production reached 413.8 million metric tonnes in 2023, and projections indicate that cumulative production could reach 33 billion tonnes by 2050 (Rios Mendoza et al., 2021). Unfortunately, only 9% of plastic waste is recycled, 12% incinerated, while the remaining 79% is either sent to landfills or disposed of in the environment (Aragaw, 2021; Kumar et al., 2021). If current production rates and plastic waste management practices persist, it is estimated that 1.2 billion metric tonnes of plastic waste might end up in landfills and the environment by 2050 (McGinnis, 2022).

Plastic pollution results from large, visible plastic items, such as plastic bags, bottles, containers, packaging materials, and fishing nets, which accumulate in the environment (Aragaw, 2021; Cristaldi et al., 2020; Rochman, 2018). In these environments, plastics undergo mechanical, chemical, photo-, and biodegradation by organisms, ultimately breaking down into small-sized plastics, referred to as microplastics (Alimi et al., 2021; Wagner & Lambert, 2018). Frias & Nash (2019) define microplastics as solid, water-insoluble particles and synthetic polymers with a size range of 1 µm to 5 mm. These tiny yet harmful substances serve as carriers of persistent organic pollutants, including heavy metals, plasticisers, antibiotic-resistant genes, polychlorinated biphenyls (PCBs), and pathogens (Sol et al., 2020; Sun et al., 2019; Wang et al., 2021). Furthermore, microplastics themselves pose a toxicity risk due to the additives and monomers used during their production, which are intended to enhance their properties, but can be harmful to both humans and ecosystems (Razeghi et al., 2021; Xie et al., 2020). For example, polybrominated diphenyl ethers and phthalates are commonly added to improve flexibility and fire resistance (Blair et al., 2019; Carr et al., 2016; Sun et al., 2019). However, these additives are also recognised as endocrine-disrupting compounds (Alvim et al., 2020; Sun et al., 2019). The small size of microplastics allows organisms to ingest them more easily compared to larger plastics, potentially acting as stressors and harming organ systems (Klein et al., 2015). Harmful compounds leached from microplastics could ultimately reach humans through the food chain via bioaccumulation and biomagnification, posing significant health risks (Aragaw, 2021).

Multiple studies have demonstrated that microplastics and associated contaminants are accumulating in both aquatic and terrestrial environments worldwide (Alimi et al., 2021; Eriksen et al., 2013). These studies provide evidence that organisms can ingest and accumulate microplastics, resulting in detrimental effects (Wagner & Lambert, 2018). Despite a growing body of global research on microplastic pollution, there is limited data from African waters, especially in freshwater ecosystems (Alimi et al., 2021). With the size of Africa's freshwater bodies as well as the people and activities they sustain, the likelihood of microplastic pollution in these waters is substantial. Biginagwa et al. (2016) conducted the first research on microplastics contamination in Lake Victoria, focusing specifically on Nile tilapia and Nile perch. The recovery of microplastics from the fish digestive system provided initial evidence of microplastics in African inland freshwater, highlighting the potential hazard that microplastics pose to the environment and public health (Migwi et al., 2020). However, despite the ecological and human health implications, little is known about the abundance and spatial variation of microplastics in Nairobi's urban rivers.

This study focused on the Nairobi River basin, specifically the Nairobi, Mathare, and Ngong Rivers, which flow through heavily industrialised and densely populated areas of Nairobi City County. These rivers bear a heavy pollution load due to various human activities, including wastewater discharge from residential areas, improper waste disposal, effluent discharges from industries and wastewater treatment plants, as well as farming activities along the riverbanks (Githaiga et al., 2023; Ngumba et al., 2016). Such factors may contribute to the presence of microplastics in these rivers. The rapid and unplanned population growth in Nairobi has led to the development of informal settlements with inadequate sanitation, increased industrial development, and a shortage of waste management facilities (Bagnis et al., 2020). The water within the catchment is a vital resource for the economic livelihood of local communities living downstream and the environmental sustainability of the region. Downstream, these rivers flow through Tsavo National Park, one of the largest National parks in Kenya, which is home to a wide variety of wildlife. The Thwake dam, currently under construction downstream, is intended to supply water to the Konza metropolis. Moreover, the surrounding agricultural and residential areas support millions of Kenyans who rely on this water for domestic use, agricultural activities, and livestock purposes (Mbui et al., 2016; Ngatia et al., 2023). While previous studies have investigated various pollutants in the Nairobi River basin (Bagnis et al., 2020; Mbui et al., 2016; Ngumba et al., 2016), microplastics have not been extensively studied. However, research on microplastics has been conducted in coastal regions, highlighting the need for further investigation into the aquatic ecosystem in Nairobi.

1.2 Statement of the Problem

Plastics are widely used in various applications due to their favourable properties, including corrosion resistance, durability, lightweight, and adaptability (Habib et al., 2020; Yang et al., 2021). However, the non-biodegradable nature of plastics raises significant environmental concerns, particularly when poor management practices exacerbate the issue (Kumar et al., 2021). While there is growing awareness of microplastics and their potential threats, few studies that quantified their concentrations in freshwater environments in Africa (Wagner & Lambert, 2018). In particular, research on river systems in Kenya, including the Nairobi, Mathare and Ngong Rivers, is limited. These rivers are a critical component of the hydrological system, feeding into larger downstream ecosystems, including ecologically sensitive areas such as Tsavo National Park. The absence of localised microplastic data not only limits understanding of pollution in Nairobi Rivers but also hinders knowledge of the potential impacts on broader ecosystems that depend on these rivers.

Previous studies have indicated that anthropogenic activities significantly influence the presence of microplastics in rivers, with higher concentrations often reported in areas with high population density (Chen et al., 2020; Kunz et al., 2023; Zhang et al., 2024). In Nairobi, factors such as industrial growth, rapid urbanisation, and inadequate waste management have intensified anthropogenic pressures on river systems. However, the specific contributions of various anthropogenic activities to microplastic abundance in Nairobi's Rivers remain unquantified.

While wastewater treatment plants are intended to reduce pollution loads, their limited microplastic removal efficiencies contribute to downstream contamination (Cristaldi et al., 2020; Sun et al., 2019). This unquantified release of microplastics into Nairobi's rivers underscores the need for targeted evaluation of their abundance and spatial distribution. Currently, microplastics are not regulated by water quality standards, given the absence of microplastics guidelines from the World Health Organisation (WHO) and the National Environment Management Authority (NEMA). This lack of regulation stems from the limited data available on the extent of their discharge. Given the increasing anthropogenic pressures and lack of baseline data on microplastic pollution in Nairobi's urban rivers and wastewater systems, there is a crucial need to evaluate their abundance, composition, and spatial distribution. This is critical for informing effective management and policy interventions, particularly because the contaminated rivers are used for domestic and ecological purposes, posing a threat to both aquatic species and human health.

1.3 Objectives

1.3.1 Main Objective

The main objective of this study was to evaluate the abundance, composition, and spatial distribution of microplastics in the Nairobi Rivers and wastewater treatment plants.

1.3.2 Specific Objectives

The specific objectives of this study were to:

1) Quantify the concentration and composition of microplastics in the Nairobi, Mathare, and Ngong Rivers.

2) Determine the relationship between anthropogenic activities and microplastic abundance in the Nairobi, Mathare, and Ngong Rivers.
3) Assess the efficacy of the Kariobangi and Ruai Wastewater Treatment Plants (WWTPs) in removing microplastics.

1.4 Research Questions

1) What are the concentration levels and composition of microplastics in the Nairobi, Mathare, and Ngong Rivers?
2) What is the relationship between anthropogenic activities and the abundance of microplastics in the Nairobi, Mathare, and Ngong Rivers?
3) How efficient are the Kariobangi and Ruai Wastewater Treatment Plants (WWTPs) in removing microplastics?

1.5 Justification of the Study

Microplastics are an emerging contaminant that poses a significant threat to both aquatic life and ecosystems worldwide. Assessing the abundance, composition, and distribution of microplastics in rivers will provide vital data regarding these contaminants and determine the extent of their contamination. Additionally, evaluating the relationship between microplastics and anthropogenic activities, as well as the removal efficiency of wastewater treatment plants, will help identify pollution hotspots and pathways. Understanding these aspects is essential for developing long-term intervention and remediation strategies to minimise plastic and microplastic pollution. These findings will provide critical baseline data to inform policy development on plastic waste management, support regulatory enforcement by the NEMA, and promote sustainable urban water resource protection in Kenya. Furthermore, understanding the extent of microplastic pollution is crucial for the sustainability of livelihoods that rely on these water resources.

1.6 Scope of the Study

This study covered the Nairobi, Mathare, and Ngong Rivers, as well as the Ruai and Kariobangi wastewater treatment plants, to assess the abundance, composition and spatial distribution of microplastics in both natural and treated water systems. Water samples from the rivers were collected in July and August 2024, while wastewater samples were collected in March and April 2025, all during the dry season. Laboratory analyses were conducted at the Chemistry and Zoology laboratories of the Jomo Kenyatta University of Agriculture and Technology (JKUAT), with additional analyses performed at the Chemistry Laboratory of the University of Nairobi (UON). All the requisite permissions were obtained from the relevant authorities. Research permission was granted by the National Commission for Science, Technology, and Innovation (NACOSTI), as detailed in Appendix 1, and permission to collect wastewater was provided by the Nairobi City Water and Sewerage Company (NCWSC), as noted in Appendix II.

1.7 Limitations of the Study

This study was conducted over a short sampling period, which may have overlooked seasonal variations in microplastic concentrations and composition. As a result, the study might have underestimated microplastic concentrations that occur during the rainy season. Furthermore, wastewater samples were collected only from the influent and effluent stages, excluding intermediate treatment stages where significant microplastic removal could occur. These limitations affect the ability to assess removal efficiency at each treatment stage and may influence the interpretation of the overall performance of the WWTPs.

CHAPTER TWO

LITREATURE REVIEW

2.1 General Overview

The term microplastics typically refers to plastic particles with a maximum diameter of less than 5 mm, as defined by most authors (Browne et al., 2011; Duis & Coors, 2016; Frias & Nash, 2019). Although the lower size of microplastics is not standardised, the mesh size of the neuston net, typically 333 µm, is conventionally used for sample collection (Duis & Coors, 2016). However, there are proposals to refine this definition, with suggestions to redefine microplastics as particles greater than 1 mm, while introducing the terms nanoplastics, mesoplastics, and macroplastics as shown in Table 2.1. While these thresholds are commonly used, variations exist among studies, indicating that microplastic classification is an evolving field. Size boundaries are continuously refined as research progresses and standardisation efforts improve.

Table 2.1: Classification of Plastics by Particle Size

Plastic Category	Size
Megaplastics	>1 m
Macroplastics	25 mm-1 m
Mesoplastics	5 mm-25 mm
Microplastics	1µm-5 mm
Nanoplastics	<1µm

Source: Browne et al. (2011); Duis & Coors (2016); Frias & Nash (2019).

Microplastics in surface water resources primarily originate from several sources, including wastewater treatment facilities, abandoned plastic fishing equipment, industrial and commercial activities, informal settlement wastewater discharge, storm drains, farmland drainage, and personal hygiene products (Horton et al., 2017; Kumar et al., 2021; Su et al., 2020). These microplastics are carried from emission sources to aquatic environments through various means, such as surface runoff, wind erosion, atmospheric fallouts, domestic and industrial drainage systems, and discharge from wastewater treatment plants, as well as from direct human activity near riverbanks (Gutiérrez-Rial et al., 2024; Mintenig et al., 2020; Zhang et al., 2024). However, the relative contributions of these sources to microplastic pollution and the mechanisms governing their transport remain unclear, particularly under local conditions (Su et al., 2020). For instance, in the Nairobi River basin, a better understanding of local microplastic sources is needed to effectively address the primary pollution sources.

Microplastics can be categorised into two classes based on their production method: primary and secondary (Andrady, 2017; Cole et al., 2014). Primary microplastics are microscopic synthetic plastic particles intentionally produced and discharged into the environment through industrial and household activities such as runoff, spills, sewage discharge, and effluents (Cole et al., 2014). These particles assume different forms, such as granules, pellets, fibres, or needles, and commonly originate from products like facial scrubs, toothpastes, and abrasive blasting materials (Dris et al., 2018; Wagner & Lambert, 2018). Secondary microplastics, on the other hand, are produced from larger pieces of plastic that already exist in the environment and which degrade to create microplastics. This breakdown can occur due to various causes, such as wave abrasion, photo-oxidation, or biological deterioration by microorganisms (Razeghi et al., 2021; Thompson et al., 2009). Environmental conditions influence the rate and pattern of degradation experienced by microplastics in ecosystems. While the use of stabilisers during plastic production helps minimise photo-oxidative degradation, fluctuations in heat conditions, as well as exposure to water or sediment, can facilitate disintegration (Duis & Coors, 2016).

Microplastics originating from both primary and secondary sources exhibit diverse variability in chemical composition, shape, colour, density, and size (Duis & Coors, 2016). Among the different types of polymer, the most often encountered in microplastics include polypropylene, polyester, polystyrene, and low-density polyethylene, as well as high-density polyethylene (Duis & Coors, 2016). By shape, microplastics can be categorised into fibres, fragments, film, pellets, and foam (Razeghi et al., 2021). While pellets are round and have significant rigidity, fragments are hard particles that are irregularly shaped. Films are very slender pieces of plastic, while fibres are very slim, homogeneous strands of plastic. Foam, on the other hand, is comprised of a vast number of small bubbles (Razeghi et al., 2021). Microplastics come in many colours, which can range from transparent to yellow, white, green, red, or blue (Scherer et al., 2020; Wagner & Lambert, 2018).

Microplastics exhibit a distinctive set of characteristics, notably a high surface area to volume ratio and inherent hydrophobicity. These traits make them prone to sorption of very harmful and persistent contaminants found in the environment, such as polychlorinated biphenyls (PCBs), pharmaceuticals, plasticisers, pathogenic organisms, and heavy metals (Cristaldi et al., 2020; Sun et al., 2019; Wagner & Lambert, 2018). Therefore, they act as sources and sinks for chemical contaminants, serving as pathways for environmental pollution and bioaccumulation of pollutants (Scherer et al., 2020). Research has shown that microplastics can absorb organic pollutants with partition coefficients ranging from 10^4 to 10^6 (Andrady, 2017; Migwi et al., 2020). These pollutants can be released inside organisms' bodies under conditions of lower pH, higher temperatures, and the presence of digestive fluids, significantly accelerating the desorption rate compared to water (Bakir et al., 2014).

Given these diverse physical and chemical properties, understanding microplastic characteristics is essential to assessing their environmental behaviour and potential impacts in Nairobi's urban rivers, where both primary and secondary sources are likely to coexist.

2.2 Microplastic Concentration and Composition in Freshwater Ecosystems

2.2.1 Microplastic Abundance

Microplastics presence has been detected in various aquatic environments globally, including lakes (Biginagwa et al., 2016; Migwi et al., 2020), estuaries (Kazour et al., 2019; Kerubo et al., 2021; Yonkos et al., 2014), rivers (Kundu et al., 2022; Razeghi et al., 2021; Zhang et al., 2024), and the ocean (Browne et al., 2011; Kosore et al., 2018). These microplastics have been found in both water surfaces and sediments. Studies have shown that the abundance of microplastics in these systems varies widely, ranging from almost zero to numerous tens of thousands of items per litre or per square meter or cubic meter, depending on the sampling method (Dris et al., 2018). Microplastic abundance varies due to several factors, including anthropogenic activities (Kundu et al., 2022; Yang et al., 2021), water catchment size (Migwi et al., 2020; Yang et al., 2021), sewage disposal methods, and the type of waste management (Habib et al., 2020; Leslie et al., 2017).

Microplastic pollution in river systems has only emerged recently and remains limited in some countries. The majority of research on microplastics in rivers has been conducted in Europe, Asia, and North America, where high concentrations of microplastics have been reported (Crew et al., 2020; Kumar, 2022). Asia, particularly China, significantly contributes to the high levels of microplastics in these environments due to its large-scale plastic production (Wang et al., 2017). Consequently, microplastic pollution in China has received considerable attention and has been extensively studied. In contrast, there are currently few studies on the state of microplastic pollution in Africa, although evidence of microplastics in African aquatic environments has been reported (Kundu et al., 2022; Weideman et al., 2020). Notably, there is little to no known published data on microplastics pollution in Kenyan river systems. However, research has been conducted on marine ecosystems (Kerubo et al., 2021; Kosore et al., 2018) and lakes (Migwi et al., 2020) as shown in Table 2.2. These studies indicate that marine ecosystems have significantly higher concentrations of microplastic compared to lakes such as Lake Naivasha, suggesting variations in source inputs and hydrodynamics. These findings highlight the need for further research into the abundance of microplastics in freshwater environments and river systems in Kenya.

Table 2.2: Microplastic Concentration in Lakes and Oceans Studied in Kenya

Source	Range	Reference
Lake Naivasha	0.183 ± 0.017 to 0.633 ± 0.067 particles/m^2	Migwi et al. (2020)
Tudor, Mida, and Port-Reitz creeks	3365.1 ± 431.2 and 2539.3 ± 223.5 MP/m^3	Kerubo et al. (2021)
Indian Ocean	110 particles per m^3	Kosore et al. (2018)

Microplastic abundance varies due to several factors, including anthropogenic activities near the rivers (Kundu, 2021; Yang et al., 2021), proximity to densely populated urban centers (Eriksen et al., 2013; Horton & Dixon, 2018), and water catchment size (Migwi et al., 2020; Yang et al., 2021). These factors contribute to abundance of microplastics due to their proximity to point and non-point sources (Yang et al., 2021). Additionally, the hydrology of the river, including factors such as flow velocity, water level, and seasonal fluctuations, can influence the concentration of microplastic particles (Liu et al., 2021; Yang et al., 2021). Weakened hydrodynamics or lower flow rate may facilitate the accumulation of microplastics, resulting in decreased concentrations in the surface water (Talbot & Chang, 2022). Sediments that accumulate higher amounts of microplastics can be re-suspended or flushed out due to increased river discharge, which can lead to a higher presence of microplastics in surface water (Su et al., 2020; Zhang et al., 2024). Studies have shown that microplastic abundances are generally lower at upstream sites compared to downstream sites, a difference attributed to variations in river discharge (Campanale et al., 2020; Talbot & Chang, 2022).

2.2.2 Microplastic Composition

Microplastics in the environment exhibit a diverse array of shapes, densities, colours, and polymer types (Yang et al., 2021). These characteristics have been extensively studied and reported in different aquatic environments, showing significant variation among rivers worldwide (Dris et al., 2018; Klein et al., 2015). The abundance of microplastics in the riverine environment is closely linked to these characteristics (Yang et al., 2021).

The shapes and sizes of microplastics vary based on their original form, the fragmentation processes, and their time of residence in the aquatic environment (Andrady, 2017; Yang et al., 2021). Microplastics are primarily classified into 5 shapes: fibres/lines/filament, fragments, foams, films/sheet, granules and pellets/beads (Mutuku et al., 2024). Fragments and fibres are the most commonly found shapes in freshwater surface water (Migwi et al., 2020; Weideman et al., 2020). These shapes predominate because they often originate from plastic degradation, and the effluents of laundry and wastewater treatment plants (Eriksen et al., 2013; Wagner & Lambert, 2018). In Nairobi Rivers, which flow through densely polluted urban areas and receive discharges from informal settlements and industries, fibres and fragments are expected to be the most abundant types of microplastics. However, other studies have found variations in the common shapes of microplastics. For instance, He et al. (2020) identified films as the most frequently occurring shape in the Brisbane River, followed by fragments and fibres. Additionally, Moore et al. (2011) identified granules as the second most common shape in the Los Angeles River, attributing this to the proximity of a plastic processing plant.

Microplastics, as observed across various studies, come in a range of colours due to the diversity of plastic products manufactured (Kerubo et al., 2021; Liu et al., 2021). However, there is no standardized scheme for colour designation of microplastics. Typically, the 12 basic colour terms from the Inter-Society Colour Council National Bureau of Standards (ISCC-NBS) are used for this purpose (Mutuku et al., 2024). The importance of colour in microplastic research is highlighted because most coloured microplastic particles are often consumed by aquatic organisms, which misinterpret them as food particles (He et al., 2020). Additionally, the colour of microplastics can provide information about their origin. For instance, transparent microplastics typically originate from products such as single-use cups, bottles, and bags, which have limited lifespans (Cole et al., 2014). In contrast, the coloured microplastics originate from a broad range of durable consumer products (Andrady, 2017). However, colour is not a reliable indicator for deducing the specific origin or type of microplastics, as it changes during sample preparation due to bleaching or during residence time in water (Wagner & Lambert, 2018). Studies have reported varieties of colours in the microplastics recovered. For instance, in a study in the Oftane River, Campanale et al. (2020) identified the most common microplastics as transparent particles (56%), followed by black microplastics (5%), and the least common were coloured microplastics (11%). Migwi et al. (2020) identified that 81% of microplastics were coloured, including 38.6% black/brown, 16.4% blue, 22.8% red/pink, 7.6% green, and 14.6% white/translucent. However, in many studies, colour particles are just described as coloured without specifying the exact shades.

Plastics are artificial polymers composed of various compounds, each imparting unique properties (Andrady, 2017). These polymers affect the degradation rate and buoyancy of microplastics, which in turn determine their transportation behaviour (Dris et al., 2018). In freshwater ecosystems, microplastics are mostly comprised of polyethylene (PE), followed by polypropylene (PP) and polystyrene (PS). The other types of polymer are less frequent and include polyethylene terephthalate (PET), polyamide (PA), and polyvinyl chloride (PVC). For instance, Dris et al. (2018) found that PP, PE, and PS were the dominant types of microplastics sampled. Similarly, Klein et al. (2015b) reported PS(29.7%) and PP(16.9%) in their study. PE, PS and PP are dominant because these polymers are found at relatively low densities, which range from 0.85 to 1.42 g/cm^3 (Razeghi et al., 2021). Polymers like PET, PVC, and nylon are comparatively denser and tend to sink (Horton & Dixon, 2018; Kumar et al., 2021). However, this density is influenced by factors such as bio-fouling, sedimentation, and re-suspension processes (Kundu et al., 2022; Razeghi et al., 2021). Scherer et al. (2020) found buoyant PE and PP in sediments, suggesting that bio-fouling had increased the particles' density, causing them to sink. Additionally, high-density polymers have been observed in surface waters due to re-suspension (Kazour et al., 2019). Variations in flow velocities, turbulence, and extreme flooding events can also lead to re-suspension (Klein et al., 2015).

Table 2.3 presents the average concentrations of microplastic pollution in various rivers worldwide, along with the different compositions identified.

Table 2.3: The Abundance of Microplastics from Various Rivers Across the World

Continent	Country	Location	Microplastics Concentration	Polymer Types	Shape of microplastics	Method of collection	References
Europe	Italy	Oftanto River	0.9 ± 0.4 p/ m^3 to 13 ± 5 particles/m^3	PE, PA, PP, PET	Foam, fibre, fragment, pellet	333mm plankton nets	Campanale et al. (2020)
	Germany	River Elbe	0.88 to 13.24 particles/m^3	PS, PE, PET, PVC	Film, fibre, fragment, , sphere	150 µm plankton net	Scherer et al. (2020)
	Germany	Rhine-main river	892,777 particles/ Km^2	PP, PS, PVC, PEST	fibre, sphere Fragment	Stainless steel spoon (30 cm^2 area)	Klein et al. (2015b)
	France	Marne river	398 particle/litre	N/A	fibre	Manta trawl of 330 µm and plankton net of 80 µm	Dris et al. (2018)
	France	Seine river	108 particles/m^3	Not detected	fibre	80 µm Plankton net	Dris et al. (2018)
Asia	China	Manas River	10–22 particle/litre	PP, PE, PS, PET,PVC	Fibre, film, fragment,	Stainless steel drums (2.5L)	Wang et al. (2021)
	China	Qing River	0.1–0.5 particle/litre	PS, PE, PP, PVC	Microbeads, fibre, fragment, film, pellet	Manta net (60 cm x 18 cm)	Wang et al. (2020)
	China	Wuhan river	1660.0 ± 639.1 to 8925 ± 1591 particles/m^3	PE, PP, Nylon, PET	Fibre, film, pellet, granule	20cm diameter Teflon pump	Wang et al. (2017)
	China	Hanjing river	2933 ± 305.5 particles/m^3	PP, PE, PET	Fibre, film, granule	Stainless steel bucket (20L)	Wang et al. (2017)
	China	Yangtze river	2516.7 ± 911.7 items/m^3	PS, PE, PP, PET, PA	Fibre granule, pellet, film	Water volume of 20L pumped over 50µm sieve	Wang et al. (2017)
Africa	Tanzania	Themi River	0.57 ± 0.27 items/ litre	PS, PE,	Fibres, fragments,	metallic jar	Kundu et al. (2022)
	South Africa	Orange-Vaal River	0.23 ± 0.27 particle/ litre	Not detected	Fibre fragment	neuston net	Weideman et al. (2019)
North America	USA	Los Angeles river	13.7 particle/litre	PS	Fragment, foam, pellet, film	333 µm Manta Trawl	Moore et al. (2011)
	USA	San Gabriel River	0.6 particle/litre	PS	Fragment, foams pellets	333 µm Rectangular net	Moore et al. (2011)
	USA	Chicago, Illinois, Mississippi Rivers	1.94–17.93 particles/m^3	Not detected	Fragment, pellet, fibre	Neuston nets of 333 µm mesh size	McCormick et al. (2016)
	Canada	St Lawrence river	0.16 ± 0.02 particles/litre	Not detected	Fibres, microbeads fragments	Using ethanol	Crew et al. (2020)

2.3 The Influence of Anthropogenic Activities on Microplastic Abundance

Plastics are a significant riverine contaminant and are considered indicators of the Anthropocene (Wagner & Lambert, 2018). The concentrations of microplastics in riverine ecosystems are influenced by various human activities, including improper waste disposal, wastewater discharge from residential areas, plasticulture farming along rivers, effluent discharges from industries, and wastewater treatment plant discharges (Eerkes-Medrano et al., 2015; Horton & Dixon, 2018). Studies have documented higher microplastic abundance in densely populated areas with intensive human activities (Chen et al., 2020; Grbić et al., 2020; Kunz et al., 2023; Yonkos et al., 2014). However, other studies have found no clear relationship between land use patterns and microplastic abundance (Barrows et al., 2018; Talbot & Chang, 2022; Wang et al., 2021). Therefore, while microplastic abundance is linked to anthropogenic activities, some studies suggest otherwise, highlighting the need for further studies to clarify this relationship. Differences between studies may be attributed to the scale of the study area and local factors (Kunz et al., 2023; Talbot & Chang, 2022). Most previous research on the relationship between land use types and microplastics has been conducted at a qualitative level (Feng et al., 2020), with few quantitative analyses available (Wang et al., 2021).

Studies on microplastic pollution indicate that the type and concentration of microplastics are highly correlated with urbanisation. Urban water bodies receive microplastics from various sources, including road runoff, industrial wastewater, indiscriminate waste dumping, domestic waste discharge, and atmospheric deposition (Eriksen et al., 2013; He et al., 2020). Numerous studies have suggested a positive correlation between microplastic quantities and urbanised or densely populated regions. For example, Yonkos et al. (2014) identified a positive correlation between microplastic pollution and the percentage of industrial areas and urban development in estuarine rivers in the Chesapeake Bay in the USA. Chen et al. (2020) also found that increased urbanisation and higher human population density within a catchment area lead to greater microplastic abundance in nearby receiving waters. The Nairobi Rivers flowing through dense urban and industrial areas may experience significantly elevated microplastic concentrations due to urbanisation. However, other studies have not found a significant or positive relationship between urbanisation and microplastic concentration, as noted by He et al. (2020). Talbot & Chang (2022) suggested that this could be due to the local regulations and waste management strategies that limit pollution.

Microplastic concentration has also been found to correlate with agricultural land use activities. Agricultural lands contribute to microplastic pollution caused by the usage of sludge as fertiliser, which reintroduces microplastics into river systems (Horton & Dixon, 2018; McGinnis, 2022; Talbot & Chang, 2022). Additionally, the use of plastic materials, such as irrigation pipes, mulch films, silage wraps, nursery tree seedling bags, and greenhouse coverings, can also contribute to microplastic pollution (Grbić et al., 2020). Studies have identified potential sources of microplastics from agricultural activities in the catchment areas. For instance, Zhang et al. (2024) linked the films and fragments detected in the Wei River to the weathering and breakdown of plastic mulch from agricultural fields. Similarly, Tan (2025) noted a prevalence of films in specific sampling points associated with agricultural regions that heavily utilise films in their production processes.

Studies have reported a decrease in microplastic concentrations in areas farther from intense anthropogenic activities, such as in forested regions and natural reserves. These areas tend to have less intensive human activities, resulting in fewer potential sources of microplastics (Talbot & Chang, 2022). They are also associated with lower population density, less development, less impermeable surface, and fewer drainage systems (Yonkos et al., 2014). In Nairobi, for instance, areas located farther from the city centre, such as undeveloped regions and forests, are expected to have lower concentrations of microplastics. For example, Townsend et al. (2019) found a negative correlation between microplastics and less developed areas in the Greater Melbourne Region, Victoria, Australia. Similarly, Yonkos et al. (2014) observed a negative relationship between microplastics and forested land use. However, these regions have been found to still contain microplastics, which is attributed to tourism and recreational activities (Talbot & Chang, 2022).

These factors contribute to the abundance of microplastics due to their proximity to point and non-point sources (Yang et al., 2021). However, elevated levels of microplastics have also been found in rivers in isolated locations lacking these point and non-point sources (Klein et al., 2015). This phenomenon is attributed to atmospheric deposition, long water residence time, and small catchment sizes (Rochman, 2018). Additionally, the hydrological condition of the river, including factors such as flow velocity, water level, and seasonal fluctuations of water flow, may amplify or mitigate the anthropogenic effects (Liu et al., 2021; Yang et al., 2021).

2.4 Efficacy of Wastewater Treatment Plants in Removal of Microplastics

Industrial and municipal effluents contain both micro and macro-plastics (Murphy et al., 2016). Although larger plastics are typically recovered in water treatment units, the current wastewater treatment plant technologies are not designed to remove microplastics from the effluents (Browne et al., 2011; Cristaldi et al., 2020). The effectiveness of WWTPs in removing microplastics varies depending on the specific treatment methods used and the operating condition of the wastewater treatment plant (Conley et al., 2019; Cristaldi et al., 2020; Wang et al., 2020), with reported removal rates ranging from 40% and 99.9% (Cristaldi et al., 2020; Sun et al., 2019).

Wastewater treatment typically involves several stages: pre-treatment, primary treatment, secondary treatment, and tertiary treatment (Habib et al., 2020). During pre-treatment, screening, grit and grease removal techniques are used to maximise microplastic removal efficiency. Microplastics can be captured if other materials block the screens, while the lightweight microplastics can be reduced through skimming in the grease removal stage (Cristaldi et al., 2020; Mintenig et al., 2017; Murphy et al., 2016). In primary treatment, the removal of microplastics is primarily achieved through skimming of buoyant microplastics and settling of high-density microplastics in primary clarifiers (Cristaldi et al., 2020; Liu et al., 2019; Mintenig et al., 2017). This preliminary treatment typically achieves an approximate removal efficiency of 35-95% and 50-98% in the primary treatment (Cristaldi et al., 2020; Habib et al., 2020).

Secondary treatment uses microorganisms to remove organic materials (Habib et al., 2020). During this stage, bacterial extracellular polymers or sludge flocs aid in the accumulation of microplastics, which can be separated from the effluent through settling in the secondary settling tanks of the treatment system (Carr et al., 2016). Microplastics have been reported to reduce to 0.2%-14% at this stage (Cristaldi et al., 2020). Before the treated water is released into a nearby water body, tertiary treatment is performed in some wastewater treatment plants. The removal of microplastics at this stage depends on the treatment technology, with membrane bioreactors showing the highest performance (Cristaldi et al., 2020; Talvitie et al., 2017). Microplastics are reported to be reduced to 0.2%-2% at tertiary treatment (Cristaldi et al., 2020; Habib et al., 2020). However, even with tertiary treatment, microplastics are not entirely removed from the final effluent, indicating the need to develop additional removal mechanisms.

Earlier research on WWTP technologies has demonstrated that these technologies are ineffective in eliminating microplastics from treated water (Browne et al., 2011; Murphy et al., 2016). The concentrations of microplastics in treated wastewater vary significantly, ranging from 0.001 items per litre in Germany (Mintenig et al., 2017) to 91 different items per litre in the Netherlands (Leslie et al., 2017) (Table 2.4). These variations can be attributed to multiple causes, including differences in treatment facilities and their conditions, the size of the WWTPs, the population served by the WWTP, temporal factors such as seasonal changes, and various sample collection and processing techniques (Cristaldi et al., 2020; Dris et al., 2018; Habib et al., 2020). As the complete removal of microplastics is not achieved, the small amounts of microplastics released, coupled with the large volumes of treated wastewater, contribute significantly to environmental pollution (Fan et al., 2023; Sol et al., 2020). For instance, Murphy et al. (2016) reported an estimated daily discharge of 65 million microplastic particles in a wastewater treatment plant in Scotland, UK.

Substantial quantities of microplastics bypass wastewater treatment plants and enter aquatic ecosystems, ultimately accumulating in the environment (Cristaldi et al., 2020; Fan et al., 2023b; Franco et al., 2021; Habib et al., 2020). However, to date, a small number of studies have examined the effectiveness of wastewater treatment facilities in eliminating microplastics and the amount of microplastics present in wastewater sludge and effluents in Africa’s wastewater treatment plants (Habib et al., 2020; Murphy et al., 2016). The absence of data hampers knowledge of the potential environment and public health effects of these wastewater systems and hinders efforts to improve them and minimise the microplastic pollution load (Cristaldi et al., 2020).

Table 2.3 presents the microplastic quantities found in the influents and effluents of WWTPs in different countries, along with their removal efficiencies. Across the reviewed studies, most wastewater treatment plants achieved removal efficiency above 80%. However, significant quantities of microplastics still escaped into the receiving waters. This reinforces the need for further research in WWTPs to identify the technologies required to optimise the removal of microplastic pollution in these systems. Nairobi’s major wastewater treatment plants, the Ruai and Kariobangi WWTPs, utilise different treatment technologies, which should be studied to assess their effectiveness in removing microplastics.

Table 2.4: Microplastic Abundance and Removal Efficiency in Wastewater Treatment Plants

WWTP Location	Treatment type	WWTP number	Sample volume (L)	Influent concentration (Mp/L)	Effluent Concentration (Mp/L)	Removal Efficiency (%)	Particles discharged (millions/day)	References
Glasgow, Scotland UK	Secondary	1	30-50	15.70 (±5.23)	0.25 (±0.04)	98.41%	65	Murphy et al. (2016)
Vancouver, Canada	Secondary	1	30	31.1±6.7	0.5±0.2	98.3%	32–97	Gies et al. (2018)
Wuhan,, China	Secondary	1	10	79.9±9.3	28.4±7.9	64.4%	-	Liu et al. (2019)
Saxony, Germany	Tertiary	12	390-1000	-	0.1-10.05	97%	90-400	Mintenig et al. (2017)
Vancouver Canada	Secondary	1	4-30	57.6±12.4	1.0±0.4	98.3%	1×10^7	Lares et al. (2018)
Beijang, ,China	Secondary	1	30	12.03 ± 1.29	0.59 ±0.22	95%	559	Yang et al. (2019)
Cartagena, Spain	Secondary	2	221.45	4.40 ± 1.01	0-2.67 & 0-5.28	90.3%	-	Bayo et al. (2020)
Xiamen, China	Secondary	7	2.98-142.98	6.55	0.59±0.49	90.52%	650	Long et al. (2019)
Finland	Tertiary	4	0.4-1000	6.9	0.005-0.3	40-99.9%	1.4-1.7	Talvitie et al. (2017)
Mersin bay, Turkey	Secondary & Tertiary	3	30	1.5-3.1	0.6-1.6	48-73%	277.3	Akarsu et al. (2019)
Eastern China	Tertiary	1	200	0.28±0.02	0.13±0.01 & 0.05±0.01	53.6% & 82.1%	6.5 & 3.5	Lv et al. (2019)
Scotland	Tertiary	1	25	3-10	<1-3	96%	12	Blair et al. (2019)
Netherlands	Tertiary	7	2	68–910	9-91	72%	-	Leslie et al. (2017)
Italy	Tertiary	1	30	2.5±0.3	0.4 ±0.1	84%	160	Magni et al. (2019)
California,,USA	Tertiary	1	189,000-232,000	1	0.00088	99.9%	0.93	Carr et al. (2016)

2.5 Microplastic Analysis Methods

2.5.1 Microplastic Sampling

Microplastics present challenges due to their small size and heterogeneous distribution, making representative water sampling difficult (Eerkes-Medrano et al., 2015; Wang & Wang, 2018). Various techniques are available for sampling microplastics, including volume-reduced and bulk water sampling (Aragaw, 2021; Eriksen et al., 2013). Bulk sampling involves collecting the entire sample without any reduction, such as through grab sampling and pump sampling (Wang & Wang, 2018). However, this method may limit the representativeness of the sample due to the relatively small amount collected (Aragaw, 2021). On the other hand, volume-reduced sampling methods, such as plankton nets and neuston nets, involve reducing the volume of water during sampling, thereby allowing for the coverage of large quantities or areas (Razeghi et al., 2021). However, rapid filtration during volume reduction may result in the discarding of a significant portion of the sample, including microplastics, especially those with a mesh size finer than that of the sampling tool (Wang & Wang, 2018).

The identification of suitable sampling methods is dependent on several variables, including the sampling matrix, the quantity to be sampled, the extent of sampling, and the equipment available (Alimi et al., 2021; Razeghi et al., 2021). For this study, grab sampling using buckets was employed to collect water samples for analysis in the field. To account for limited coverage and representativeness, multiple replicates were conducted. Additionally, to improve representativeness, a large sample volume is collected over space and time (Prata et al., 2019; Rios Mendoza & Balcer, 2019). Water volume can vary from 5 mL to 500 L, depending on the study. However, achieving high volumes may require time-consuming processes, such as site filtration or sieving using a bucket (Razeghi et al., 2021).

2.5.2 Sample Preparation for Microplastics Analysis

After collecting samples, it is necessary to extract and purify microplastics to eliminate impurities before quantifying and characterising them (Aragaw, 2021). To separate microplastics from organic and inorganic particles, several techniques have been developed, including sieving, organic digestion, density separation, and microfibre filtration (Eriksen et al., 2013; Wang & Wang, 2018). Sieving is a frequently employed technique for extracting microplastics from sediment and water. Stainless steel sieves are commonly utilised. These sieves trap solids bigger than the mesh size and allow smaller particles and water to pass through (Wang & Wang, 2018). The size of the mesh relies on the microplastic size ranges to be collected, which generally range from 0.035 to 4.75 mm (Dris et al., 2018).

Samples often contain high concentrations of organic compounds or biofilms attached to the plastic particles, which can lead to an overestimation of microplastic quantity and hinder identification (Dris et al., 2018). To address this, digestion methods are used to remove organic materials from polymers without compromising their structural or chemical integrity (Alimi et al., 2021). Wet peroxide oxidation (WPO) is commonly used for digesting natural organic matter, as recommended by the National Oceanic and Atmospheric Administration (NOAA) (Masura et al., 2015). This treatment is carried out with a solution of 30% Hydrogen peroxide (H_2O_2) and ferrous iron (II) sulphate ($FeSO_4$). The Wet peroxide oxidation can effectively decompose organic impurities without modifying the morphology and chemical composition of microplastics (Sol et al., 2020; Wang & Wang, 2018). Organic matter can also be effectively disintegrated by mineral acids or alkalis (Aragaw, 2021). However, strong oxidising agents like HNO_3 and HCl can damage microplastics at lower pH values (Aragaw, 2021). Alimi et al. (2021) and Aragaw (2021) found that using a 10% KOH solution effectively digests biological tissues without degrading the polymers. Additionally, enzymatic digestion processes using enzymes such as lipase, amylase, proteinase, and chitinase have been explored (Aragaw, 2021; Sol et al., 2020). While these methods are plastic-friendly, they may be costly due to the enzymes involved, and the exposure time for the removal of organic matter is very long (Dris et al., 2018; Sun et al., 2019).

Density separation separates microplastics from other inorganic materials, such as bulk sediment samples, based on their density differences. When sediments are combined with a dense salt solution, the sediment sinks to the bottom. At the same time, microplastics float on the surface and are isolated by filtration through screens, meshes, or filter papers (Sol et al., 2020). A large number of saline solutions can be used, including sodium iodide (NaI; d=1.6-1.8g/cm^3), sodium chloride (NaCl; d=1.2 g/cm^3), Zinc Chloride ($ZnCl_2$; d=1.5-1.7 g/cm^3), and sodium Poly-tungstate ($H_2Na_6O_{40}W_{12}$; d = 1.4-1.6 g/cm^3). However, high-density salts, although effective, can be costly and environmentally hazardous (Aragaw, 2021). Sodium chloride has been identified as the most reliable and popular technique for separating microplastics. This approach has also been found to be suitable for regular environmental policy monitoring because it is quick, easy to use, and does not include any dangerous chemicals. It is also often utilised due to its availability, affordability, and low toxicity for this process (Sol et al., 2020; Wang & Wang, 2018).

2.5.3 Quantification and Identification of Microplastics

The next stage in microplastic analysis constitutes the detection and quantification of microplastics that have been retrieved. As environmental samples are complex, a combination of methods is necessary, as no single method is sufficient for identifying microplastics (Alimi et al., 2021). Eriksen et al. (2013) emphasise the importance of using a reliable identification technique, as particles visually identified as microplastics may actually be aluminium silicates. The identification of microplastics typically involves visual inspection to count and determine the shapes, colours and sizes of microplastics, as well as spectroscopic techniques to assess the polymeric composition of microplastics (Wang & Wang, 2018). Scanning electron spectroscopy, pyrolysis-GC/MS, FT-IR, and Raman spectroscopy are the primary spectroscopic methods used in the analysis of microplastics, as shown in Table 2.5 (Alvim et al., 2020; Dris et al., 2018; Liu et al., 2019).

Table 2.5: Comparative Overview of Analytical Techniques Used for Characterising Microplastics

Techniques	Detection Limits	Advantages	Disadvantages
Microscope counting (Stereo or dissecting)	For large microplastics, 1-5mm size range	Simple, fast and easy Advantageous to high amount of samples hence save time and cost	Cannot determine the polymer type High possibility of false positive
μ-FTIR and ATR-FTIR	ATR-FTIR: Plastic particles greater than 500μm μ-FTIR: smaller particles less than 20μm	Samples are not destroyed Acquisition of several thousand spectra	Only effective for smaller particles less than 20μm Expensive Needs experienced researcher
Raman Spectroscopy	Suitable for 1 to 20μm sized particles	Effective for particles between 1 and 20μm which cannot with FTIR Chemical mapping is fast Have automatic data collection	The samples must be well treated Time consuming analysis
Scanning Electron Microscopy (SEM)	Nano-sized particles can be analysed	Samples are not destroyed High resolution image of the samples Give reliable polymer types of the sample	Not applicable for large number of samples Samples must be highly vacuum coated It is costly
Pyrolysis GC/MS	Suitable for only greater than 500μm size samples	Organic plastic inclusive samples can be analyzed High sensitivity and reliability Data library is available for common polymer spectra	The database is only available for selected polymers Only effective for large size particles

Microplastics are commonly identified by their visual properties, such as shape and colour (Dris et al., 2018; Hidalgo-Ruz et al., 2012). However, potential biases exist in visual identification influenced by factors such as sample matrix, examiner experience, microscope quality, and particle characteristics (Alimi et al., 2021). These biases can result in overestimation and underestimation of microplastics (Dris et al., 2018). To improve identification accuracy, Hidalgo-Ruz et al. (2012) have suggested strict selection criteria to ensure consistency in particle characteristics, which include: 1. Fibre particles having distinguishable and uniform colour and thickness through the length are counted 2. Suspected particles with no visible organic structures are counted 3. Particles with uniform and clear colour throughout their length are counted 4. Particles that are white and transparent are inspected under higher magnification. Additionally, utilising spectroscopic instruments such as FTIR, SEM, Raman, Pyr-

GC/MS, or other analytical techniques to confirm microplastic identity, especially for smaller items, is recommended to minimise error rates in visual identification results.

A Scanning Electron Microscope (SEM) is a device that uses a high-speed electron beam to scan the surface of a sample in a raster sequence, providing high-resolution images of particles as small as 5 μm (Alimi et al., 2021). By examining the surface morphology of microplastics through SEM, it is possible to differentiate them from other impurities and provide detailed composition information (Wang & Wang, 2018). However, preparing samples for SEM requires a significant amount of time and effort, and is not suitable for analysing a large number of samples (Dris et al., 2018). Pyr-GC/MS, on the other hand, analyses thermal degradation products to identify the composition of microplastics in environmental samples (Dris et al., 2018; Wang & Wang, 2018). Pyr-GC/MS enables the rapid analysis of polymer particles with minimal sample preparation, providing detailed information about the microplastic polymer and any included organic additives (Alimi et al., 2021). It is insensitive to particle size, associated contaminants, and shape, and is applicable for trace analysis. However, Pyr-GC/MS analyses only one particle per cycle, with each measurement taking 30 to 100 minutes, which limits its usefulness for large quantities of samples (Dris et al., 2018). Additionally, particles must be larger than 100 μm to be manually inserted into the pyrolysis tube (Wang & Wang, 2018).

Raman spectroscopy is a non-destructive method used for identifying the chemical composition of microplastics. The sample's molecular structure causes a varying frequency of the backscattered light when irradiated with a monochromatic laser beam via scatter, absorption, or reflection (Dris et al., 2018). Raman spectroscopy, compared to FTIR, offers narrow spectral bands, a larger spectral range, greater spatial resolution, and less sensitivity to water interference (Alimi et al., 2021; Wang & Wang, 2018). It also enables the chemical imaging of the entire sample with resolutions of less than 1 μm. However, Raman spectroscopy can be compromised by pigments, additives, or chemicals attached to microplastics, which can affect identification accuracy. Additionally, its signal-to-noise ratio is typically low, increasing the difficulty of spectral analysis. Moreover, high-resolution Raman spectroscopy requires significantly more time than micro-FTIR spectroscopy when analysing entire filters

(Dris et al., 2018). Given the scale and resource limitations of this study, ATR-FTIR was selected for its versatility and accuracy.

FTIR spectroscopy provides infrared spectra that reveal specific chemical characteristics, enabling the identification of unknown substances by comparing them with available reference materials (Alvim et al., 2020; Yang et al., 2021). FTIR can be used to scan entire suspended particles or analyse subsamples, confirming visual identification results. Due to its versatility, FTIR is a common technique for the chemical characterisation of microplastics found in environmental samples (Alimi et al., 2021). However, it is limited to particles 10-20μm in size and may not apply to smaller particles (Wang & Wang, 2018). Confirmation of microplastic identity using FTIR takes time and needs expertise in operation. To overcome this, optimised FTIR technologies such as FTIR micro-spectroscopy (micro-FTIR), attenuated total reflectance-FTIR (ATR-FTIR) spectroscopy, and focal plane array (FPA-FTIR) are increasingly being used. Micro-FTIR detects particles larger than 10 µm, while ATR-FTIR enables the analysis of larger particles, greater than 500 µm, without requiring sample preparation (Dris et al., 2018).

2.5.4 Data Reporting

Microplastics in water samples are reported in various formats and units, such as particles per unit volume, area, or mass, depending on the sample and sampling instrument used (Hidalgo-Ruz et al., 2012). The abundance of microplastic particles in water samples is typically reported based on the volume of water expressed as particles per cubic meter or particles per litre. It can also be reported based on the water surface area expressed as particles per square kilometre or particles per square meter (Dris et al., 2018; Mutuku et al., 2024). Measurement per unit area is commonly used for data collected with nets, while measurement per unit volume is mostly employed for data collected using buckets, pumps, or bottles (Yang et al., 2021). In this study, the samples were collected directly in litres. Therefore, the concentration was calculated using the formula for the number of particles per the unit volume of water sampled (Equation 2.1)

$$C = \frac{N}{V} \tag{2.1}$$

Where;

C= Microplastic concentration (particles/litre)

N= Total number of microplastic particles

V= Volume of water sampled (litres).

2.6 Summary of Literature Review and Research Gap

Microplastic pollution has been extensively documented in marine environments (Andrady, 2017; Cole et al., 2014), and significant research has also been conducted in lakes (Eriksen et al., 2013; Migwi et al., 2020). However, studies on microplastic pollution in river systems have only recently emerged and remain limited in several countries, particularly in Africa. Few studies provide quantitative data on the abundance, composition, and distribution of microplastics in these regions (Wagner & Lambert, 2018). Notably, there have been no known documented studies focusing on river systems in Kenya.

The relationship between anthropogenic activities and microplastic abundance is also poorly understood, often analysed on a qualitative level, with quantitative studies being limited (Talbot & Chang, 2022; Wang et al., 2021). Furthermore, research indicates that wastewater treatment plants (WWTPs) contribute to the release of microplastics into the environment as point sources (Crew et al., 2020; Kumar et al., 2021). Therefore, effectively detecting the occurrence and fate of microplastics in WWTPs is crucial for their control. Hence, this study aims to quantify microplastic levels across varying land uses and assess the treatment efficiency of Nairobi’s major WWTPs.

2.7 Conceptual Framework

The conceptual framework for this study illustrates the interconnection between independent, intervening, and dependent variables in evaluating the abundance, composition, and spatial distribution of microplastics in the Nairobi Rivers and wastewater treatment plants (Figure 2.2).

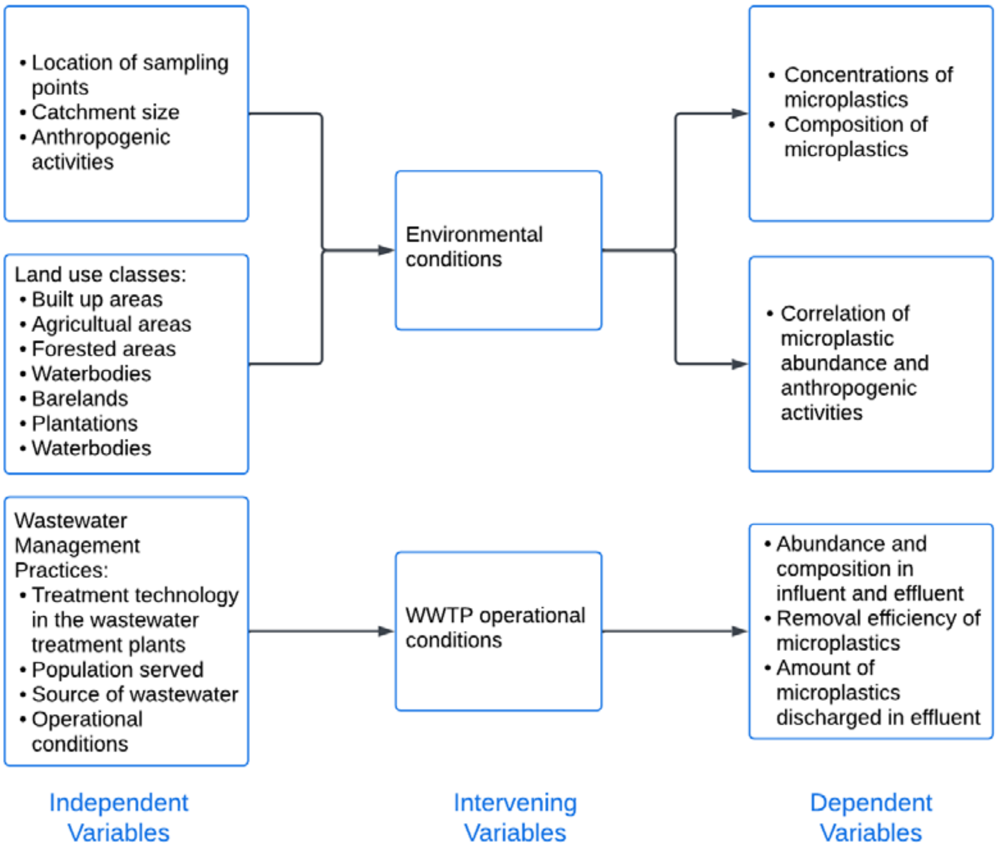

Figure 2.1: The Conceptual Framework Used to Analyse Composition and Abundance of Microplastics in Nairobi Rivers

CHAPTER THREE

MATERIALS AND METHODS

3.1 Description of the Study Area

The study was conducted in the Nairobi River basin, which encompasses an area of approximately 700 km^2, with altitudes ranging between 1,500 and 1,800m (Nyika, 2017). The study focused on the Nairobi, Mathare, and Ngong Rivers. These rivers cover a significant area of Nairobi City County, flowing southeast into the Athi River and eventually reaching the Indian Ocean. The Ngong River originates from the Motoine swamp and Dagoretti forest, flows through informal settlements such as Kibera and Mukuru, and industrial areas before merging with the Nairobi River at Njiru (Ngatia et al., 2023). The Mathare River originate from the Kabete Dam and flows through the Mathare and Korogocho slums before merging with the Nairobi River (Ngumba et al., 2016). The Nairobi River begins at the Ondiri swamp in Kikuyu, flowing through Kamukunji, Gikomba market, Shauri Moyo, and Majengo (Mbui et al., 2016). At the confluence of the Nairobi and Ngong rivers lies the Dandora dumping site, which serves as the primary waste disposal site for the entire city of Nairobi (Bagnis et al., 2020).

Sampling sites were strategically selected to ensure a representative spatial distribution across the Nairobi, Mathare and Ngong Rivers. A total of 15 sites were selected, comprising six along the Nairobi River, five along the Ngong River, and four along the Mathare River (Figure 3.1). The sampling design considered the upstream, midstream, and downstream sections of the city centre, as well as proximity to diverse anthropogenic activities and land uses within the Nairobi River Basin (Table 3.1).

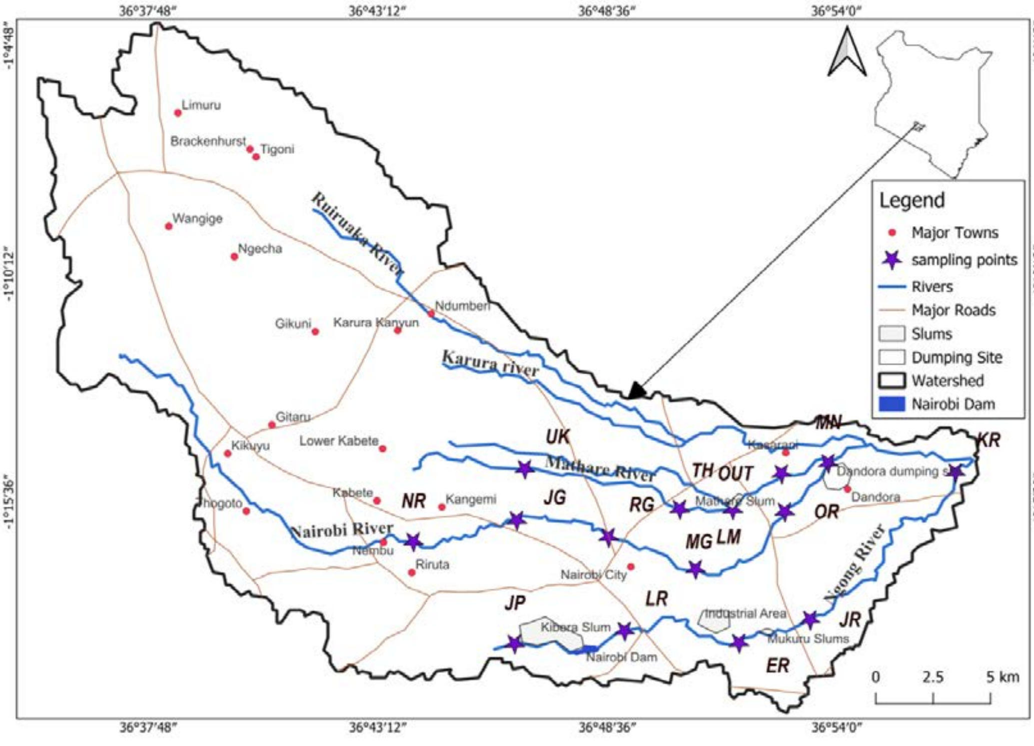

Figure 3.1: Map of the Nairobi River Basin Showing the Study Area and the Location of the Sampling Points

Table 3.1: Sampling Sites and GPS Locations at the Nairobi River Basin

ID	Station Name	Site Justification	River Section	Coordinates Latitude(ºS)	Longitude(ºE)	Chainage (km)[a]	River
JP	Jamhuri Park	Upstream of the city Centre	Upstream	-1.31799	36.77519	0+000	Ngong
LR	Langata road bridge	Downstream of Kibera Slum	Midstream	-1.31144	36.81695	4+800	Ngong
ER	Enterprise road bridge	Downstream of the industrial area	Midstream	-1.31607	36.86163	9+800	Ngong
JR	Jogoo road bridge	Downstream of informal settlements and industrial area	Midstream	-1.30621	36.88945	12+900	Ngong
KR	Kangundo road bridge	Downstream of the city centre	Downstream	-1.24886	36.94567	20+600	Ngong
NR	Naivasha road	Upstream of the city centre	Upstream	-1.27611	36.73419	0+000	Nairobi
JG	James Gichuru road	Upstream of the city centre	Upstream	-1.26728	36.77451	4+600	Nairobi
RG	Ring road	Downstream of formal settlements	Upstream	-1.27418	36.81043	8+800	Nairobi
LM	Lamu road	Downstream of Gikomba Market	Midstream	-1.28701	36.84436	12+400	Nairobi
OR	Outering	Downstream of city centre	Midstream	-1.26417	36.87960	16+300	Nairobi
MN	Confluence of Mathare and Nairobi River	Confluence of the river	Downstream	-1.24259	36.90006	18+400	Nairobi
UK	Upper Kabete	Upstream of the city centre	Upstream	-1.24727	36.77764	0+000	Mathare
TH	Thika road bridge	Upstream of Mathare slums	Midstream	-1.2631	36.83840	4+100	Mathare
MG	Confluence of Mathare and Getathuri river	Confluence of a river	Midstream	-1.26257	36.85949	8+500	Mathare
OUT	Outering	Downstream of the Mathare slum	Downstream	-1.24941	36.87835	11+100	Mathare

[a] The chainage represents the distance between the sampling points from the most upstream points.

3.2 Concentration and Composition of Microplastics in the Surface Waters

3.2.1 Surface Water Sampling

Water samples were collected in July and August 2024 from the 15 sites along the three rivers (Table 3.1). Sampling was conducted at the end of the rainy season in July and during the dry season in August. The samples were collected in triplicate, resulting in a total of 45 samples. Each sample replicate was individually treated and analysed. Samples were collected using a similar approach to that employed by Kundu et al. (2022), Jiang et al. (2019), and Ding et al. (2019), with some modifications, including the sampling volume and sieve mesh sizes. Samples were collected at each sampling point using a 20 L metallic container, where water was collected five times at the midpoint of the river at a depth of 0-30 cm from the water surface, to capture the suspended load transport of microplastics (Lofty et al., 2023). A total of 100 L was collected at each sampling point to ensure sufficient particle counts for reliable quantification (Razeghi et al., 2021). The samples were filtered in the field using a stacked stainless-steel sieve with openings of 4.75 mm and 300 µm (Hidalgo-Ruz et al., 2012; Masura et al., 2015). Particles greater than 4.75 mm were discarded, and particles between 300 µm and 4.75mm were retained for analysis. The 300 µm sieve was gently rinsed with distilled water to dislodge and transfer the retained particles into a 500 mL glass bottle. The filtered samples were then transported in a cooler box to the laboratory for further processing and analysis. At each sampling point, coordinates were recorded using a handheld GPS (Table 3.1).

3.2.2 Sample Processing- Removal of Organic Matter

Upon delivery to the laboratory, the residues collected from the 300 µm sieve were transferred to 500 mL glass beakers and oven-dried at 90 °C to sample dryness, which took 24 hours. All samples were processed according to published methods developed by the National Oceanic and Atmospheric Administration (NOAA) (Masura et al., 2015). For extraction, the particles were subjected to a wet peroxide oxidation process to digest any organic materials present. 20 mL of 30% hydrogen peroxide (H_2O_2) and 20 mL of 0.05 M Fe (II) solution were added to the glass beakers as the digester and catalyst, respectively. The mixture was then heated on a hot plate at 75 °C to oxidise any organic matter. The solution was allowed to settle in a fume hood and was covered with aluminium foil until it stopped boiling. An additional 20 mL of the H_2O_2 was added when visible organic matter was still present (Masura et al., 2015). This process was repeated until all organic materials present in the samples were fully digested.

3.2.3 Separation of Microplastics from the Samples

After the digestion of organic matter, the samples underwent density separation to isolate microplastics from inorganic materials, such as bulk sediment samples. This process was carried out using saturated Sodium Chloride (NaCl) with a density of 1.2 g/cm^3, chosen for its low cost, high availability, and environmental friendliness (Wang & Wang, 2018). 6 g of sodium chloride for every 20 ml of the sample was added and stirred with a metallic rod. The solution was heated on a hot plate set at 75 °C until the salt had completely dissolved. The mixed solution was then transferred to the density separator, and the beaker was repeatedly rinsed with distilled water to ensure all the particles were transferred to the separator. The mixture was left to settle overnight. The settled solid in the density separator was subsequently disposed of. Due to the substantial amount of inorganic matter in the samples, the density separation process was repeated three times for each sample to ensure the recovery of the microplastic particles. The supernatant solution was decanted using a vacuum pump (VE115N N Series Vacuum Pump, China) and filtered through a 0.45 µm pore size membrane filter (Chmlab, 47mm Ø, 0.45 µm, Spain). The filter was then placed in a petri dish and air-dried for further analysis.

3.2.4 Identification and Categorization of Microplastics

All particles retained in the filter were visually examined at a 40X magnification using a stereomicroscope (Optika Microscopes, Italy), following a procedure described by Hidalgo-Ruz et al. (2012) and Masura et al. (2015). Microplastics on the filter paper were collected using tweezers, then counted and categorised according to their physical characteristics. Microplastic particles were characterised by their colour (black, red, blue, brown, grey, transparent, yellow, and white) and shape (fragment, sphere, fibres, microbeads, granules, and foams). Particle identification adhered to the established guidelines by the National Oceanic and Atmospheric Administration

(NOAA) (Masura et al., 2015) and Hidalgo-Ruz et al. (2012). This included counting fibre particles that had a consistent and distinguishable colour and thickness throughout their length, as well as counting particles with no visible organic structures. Particles with a uniform and distinct colour were counted, and those that were white or transparent were inspected under higher magnification powers (40-50X). Following identification, a hot needle test was carried out on all particles identified as microplastics for confirmation. In this test, a hot needle, held with tweezers, was pointed at the suspected microplastics. The hot needle caused the plastic to become sticky and left a mark, whereas non-plastic materials did not melt or deform (Dahms et al., 2020; Ruggero et al., 2020). The hot needle test, along with established guidelines and cross-validation by a second observer, were employed in particle sorting to significantly reduce errors and biases commonly associated with visual sorting.

Due to the high number of particles counted, as well as cost and time constraints, it was not possible to identify all particles. Therefore, a random subset of 60 suspected microplastics was selected for analysis. The selection process was entirely randomised, and samples were picked from each filter using tweezers and placed in a glass petri dish for characterisation. These particles were characterised by polymer types using a Fourier transform infrared spectrophotometer (FTIR) (IRAffinity-1S FTIR device, Shimadzu, Japan). The FTIR analysis was conducted in attenuated total reflectance mode, with all spectra recorded over a wavelength range from 4000 to 400 cm^{-1}. The accumulation time was set at 1 minute with a spectral resolution of 0.5 cm^{-1}. The FTIR spectra were then subjected to a library search using the Shimadzu LabSolution IR Spectral Library (version 2.2, Shimadzu Corporation, Kyoto, Japan), as well as an open-source database, the OpenSpecy spectra analysis software (Cowger et al., 2021). A polymer's chemical composition was accepted if the overall match similarity yielded a Pearson's R-value ≥ 0.7 (Migwi et al., 2020; Wang et al., 2021).

3.2.5 Quality Assurance and Quality Control

To minimise the possibility of cross-contamination, precautions outlined by Masura et al. (2015) and Duis & Coors (2016) during sample collection, pre-treatment, and analysis processes were adopted. The precautions included rinsing all equipment with distilled water before use, wearing cotton lab coats when handling samples, covering each sample with aluminium foil to prevent airborne contamination, using glassware instead of plastic whenever possible, and cleaning all laboratory surfaces, such as benches and fume hoods, with 70% ethanol to prevent contamination (Cole et al., 2014).

To monitor for environmental and handling contamination during sample preparation, procedural blank samples of distilled water were analysed following the same methodology as sample preparation (Su et al., 2020; Wang et al., 2021). Three procedural blanks, each containing two litres of distilled water, were prepared and analysed. No microplastics were found in the blank samples, indicating negligible contamination under laboratory conditions. To validate the extraction process, 500mL of distilled water was spiked with ten polyethylene pellets, each less than 5mm in diameter (Masura et al., 2015). After pre-treatment, the microplastics captured on the filter were visually identified and counted. This step was carried out in triplicate, and a 100% recovery rate was achieved for each trial, indicating a reliable extraction process.

3.2.6 Statistical Analysis

Microplastic abundances at each sampling point were calculated based on the filtered water and reported in microplastic particles per litre. The results from the three replicates were averaged into a single value for each sampling location, and the concentrations were expressed as the mean ± standard deviation (SD). The microplastics characteristics, including shape, colour, and polymer distribution, were reported as the percentage composition of the total microplastic count per sample. The Shapiro-Wilk normality test was conducted to determine whether the microplastic concentrations were normally distributed across the sampled locations. Additionally, Levene's test was used to assess the homogeneity of variance. The results indicated that the data were normally distributed ($P=0.417$) and homogeneous ($P= 0.9714$). Therefore, a Two-Way Analysis of Variance (ANOVA) was used to compare the differences in microplastic concentrations between sampling locations and rivers, with significance accepted at $P < 0.05$. A post-hoc Tukey test was then performed to determine if there was a significant difference between the sites. All data were analysed in R statistical software (Version 4.4.2), and plots were generated using the ggplot2 package. Geodata were processed in QGIS (Version 3.36.2).

3.3 Relationship Between Microplastics Abundance and Anthropogenic Activities

A land use map for the Nairobi River basin was created to determine the correlation between anthropogenic activities and microplastic abundances. Different land use types served as indicators of varying levels of anthropogenic influences (Lin et al., 2021). For instance, built-up areas were associated with wastewater discharge from residential areas, effluent discharge from industrial areas, and wastewater treatment plants. They were also associated with road runoff and improper waste disposal from commercial and residential areas. On the other hand, agricultural land use was associated with the use of plastic mulch, irrigation with wastewater, greenhouse farming, and application of biosolids.

3.3.1 Satellite Images Acquisition

A multi-temporal Sentinel 2A imagery acquired on 9th August 2024, with a spatial resolution of 10m and with a less than 5% cloud cover, was obtained from Copernicus Open Access Hub (https://dataspace.copernicus.eu/). The image was clipped to the extent of the study areas and projected to the WGS 1984/UTM zone 37 ^{o}S map projection in the R software. The image was used to extract the land use and land cover map.

3.3.2 Land Use/Land Cover Classification

The image processing was performed using the Random Forest package in R software, following the same criteria as described by Omwoyo et al. (2024) and Kipkulei et al. (2025). Supervised training was conducted to define various land use classes, which include built-up areas (residential areas, industrial and commercial areas), forest, water bodies, agriculture (rain-fed and irrigated), plantations (a subset of agriculture, including tea plantations), bare lands (degraded lands and natural rocky terrain), and wetlands. The seven land use classes were defined based on the Food and Agriculture Organisation (FAO) land cover classification system guidelines (Gebelin et al., 2024). The training samples for the random forest classification were obtained through manual digitisation of representative polygons using the Sentinel-2 satellite image, in both true colour and false colour composite, along with high-resolution Google Earth imagery and field knowledge of the study area. A total of 225 training samples were collected, distributed across the land use classes: agriculture (n= 40), forest (n= 40), built-up areas (n= 35), bare lands (n= 25), water bodies (n= 20), plantations (n= 30), and wetlands (n= 35). The sample distribution in the training data was based on the proportional area of each land use class, with dominant classes and those with overlaps receiving additional samples to improve model accuracy. An accuracy assessment yielded an overall accuracy of 96% and a kappa coefficient of 0.94.

3.3.3 Catchment land Use/Land Cover Area

The land use map was then exported to Quantum Geographic Information System (QGIS) software (https://qgis.org/) to calculate the areas of each land use type. The catchment areas were subdivided into sub-catchments, ensuring that each sampling point was used to delineate a sub-catchment that drains into that point (Kunz et al., 2023; Tamminga et al., 2022). The individual sub-catchments were extracted from the global watersheds (https://mghydro.com/watersheds/). A visual validation of drainage boundaries was then conducted by overlaying the catchment boundaries on the Digital Elevation Model (DEM) and existing hydrological layers. In each sub-catchment, the area occupied by different land uses was calculated in QGIS, following the same criteria as Gutiérrez-Rial et al. (2024) and Kunz et al. (2023), to analyse the relationship between microplastic abundance and anthropogenic activities. The areas were then calculated as a proportion of the catchment area within each sub-catchment (Barrows et al., 2018).

3.3.4 Statistical Analysis

Pearson Product-Moment Correlation analysis was used to investigate the relationship between microplastic concentration and land use, as the data sets were normally distributed (P= 0.7454). All statistical analyses were conducted using R statistical computing software (Version 4.4.2), and Geodata were processed in QGIS (Version 3.36.2).

3.4. Efficacy of Selected Wastewater Treatment Plants in Removal of Microplastics

3.4.1 Description of the Selected Wastewater Treatment Plants

The study was done at two major wastewater treatment plants in Nairobi: The Kariobangi Sewerage Treatment Plant and the Ruai Sewerage Treatment Plant (Figure 3.2). The Ruai WWTP is the second largest wastewater treatment plant in Africa and the largest in Nairobi City (Maina, 2020). It was first commissioned in 1978 and treats both domestic and industrial wastewater. This facility employs a lagoon-based system with artificial aeration to enhance biological oxidation, as shown in Figure 3.3. It has a treatment capacity of 160,000 m^3/day, which accounts for about 80% of the wastewater produced in Nairobi (Kilingo et al., 2021; Maina, 2020). The treated effluent is discharged into the Nairobi River.

The Kariobangi Sewerage Treatment Plant, commissioned in 1963, is a conventional mechanical plant that only receives domestic wastewater. It is equipped with a biological treatment process based on biological aerated filters, as shown in Figure 3.3. This plant serves approximately 20% of the wastewater generated by Nairobi's population and has a treatment capacity of 32,000 m^3/day (Kilingo et al., 2021). The sludge is anaerobically digested to produce biogas, primarily methane, which is then utilised for the plant's energy needs. The treated effluent is discharged into the Nairobi River. The treatment processes of the two plants are illustrated in Figure 3.3.

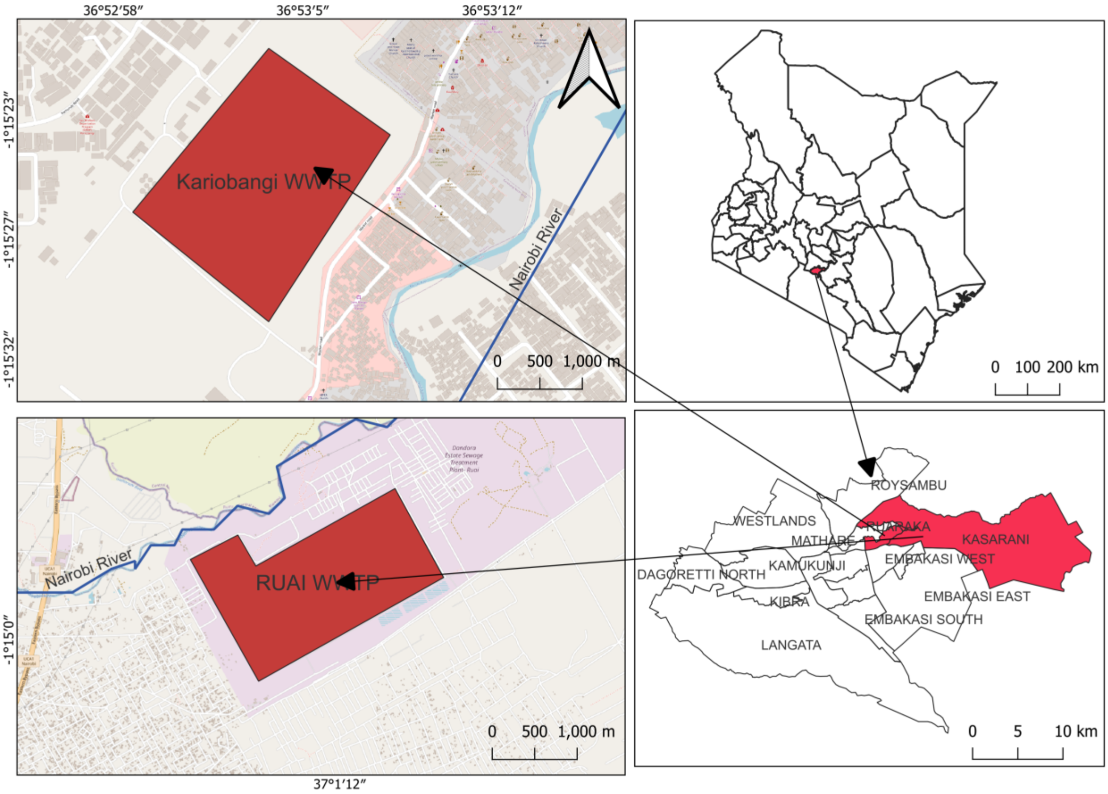

Figure 3.2: Map of Ruai and Kariobangi Wastewater Treatment Plants

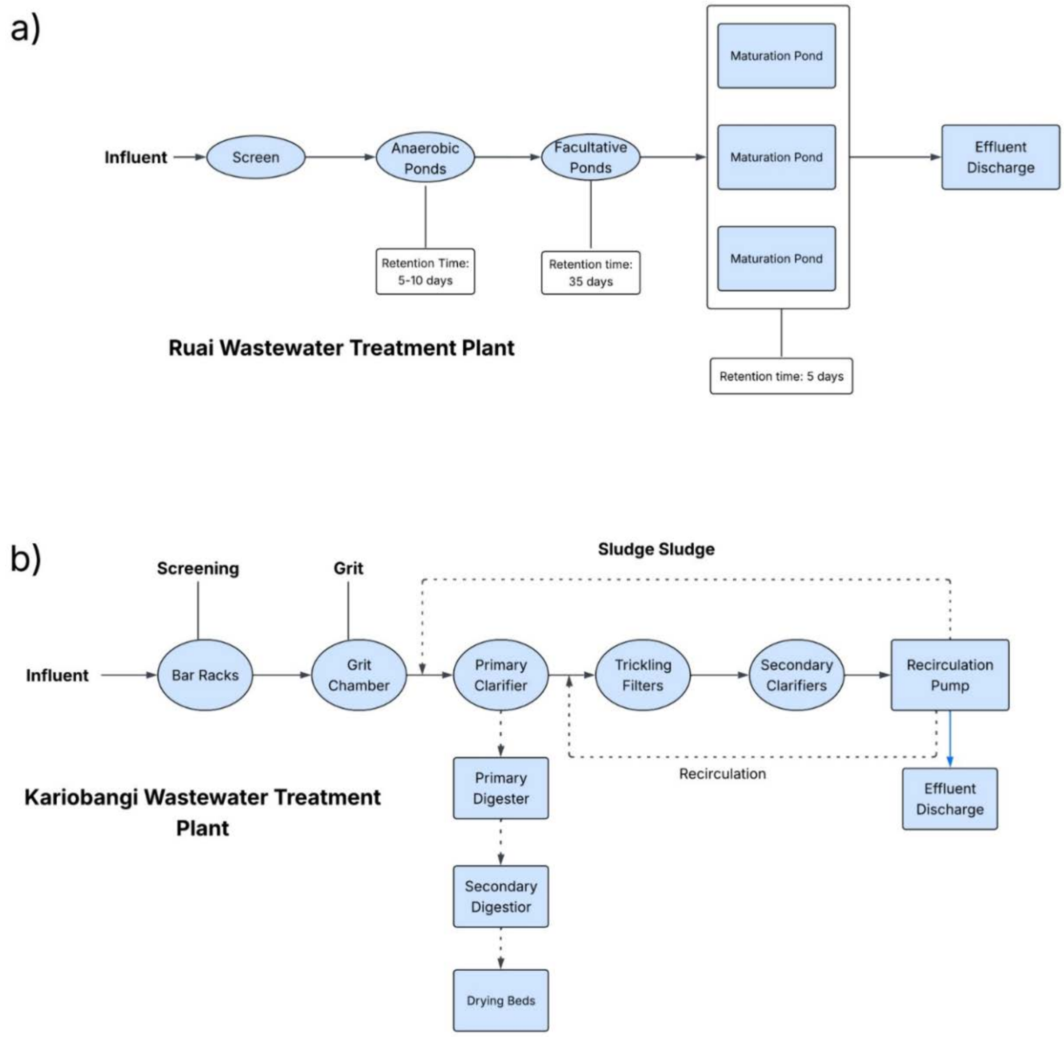

Figure 3.3: Schematic Process Flow Diagram of (a) Ruai Wastewater Treatment Plant and (b) Kariobangi Wastewater Treatment Plant

3.4.2 Wastewater Sample Collection

The wastewater sampling points were situated at the plant's influent and effluent points. Sampling took place between March 19 and April 2, 2025. The sample collection method was adapted from the approaches used by Lares et al. (2018) and Liu et al. (2019), with modifications to the sample volume and sieve mesh sizes. During each sampling event, grab samples were collected using a sampler, which consisted of a 10-litre steel bucket attached to a steel wire. Triplicate samples were taken at each stage per plant, resulting in a total of 12 samples. Influent samples were collected after coarse screening to prevent large debris from clogging and damaging the sieves. The samples were passed through stainless steel sieves with mesh sizes of 4.75 mm and 300 µm to eliminate larger debris and capture microplastic particles, respectively (Masura et al., 2015). Due to the significant amount of debris, it was only possible to filter 50 L of wastewater before the sieves became clogged. Particles greater than 4.75 mm were discarded, while those between 300 µm and 4.75 mm were retained for analysis. The residue collected on the 300 µm sieve was gently rinsed with distilled water to dislodge and transfer the retained particles into a clean 2.5 L glass bottle. The bottles were then sealed and transported in a cooler box to the laboratory for further processing and analysis.

3.4.3 Microplastic Extraction

All samples collected from the sieve were subjected to wet peroxide oxidation and density separation in the laboratory, as described in sections 3.2.2 and 3.2.3.

3.4.4 Microplastic Identification and Characterization

After the extraction of microplastics, particle characterisation involved a two-step process as described in section 3.2.4. For polymer identification, a sub-sample of 71 particles was selected and analysed using the ATR-FTIR.

3.4.5 Quality Assurance and Quality Control

To minimise the risk of contamination, several precautionary measures were implemented as discussed in section 3.2.5. To estimate contamination, three procedural blanks of 2 L of distilled water were run through the same sample equipment and processed in the same manner as the wastewater samples (Mintenig et al., 2017; Wang et al., 2020). Only fibres were detected in the blanks, with concentrations ranging from 0 to 3 fibres per blank measurement. The microplastic counts reported in this study were corrected for these blanks by subtracting the average count from the three negative controls. Positive control was conducted in triplicate with 500 mL of distilled water spiked with a known number of pellets (Long et al., 2019; Masura et al., 2015). After pre-treatment, the microplastics captured on the filter were visually identified and counted. A 100% recovery rate was achieved, indicating that the method was reliable for extracting and quantifying microplastics. No visible deformations were observed in the recovered microplastics.

3.4.6 Statistical Analysis

The total number of microplastics counted in wastewater samples was blank-corrected, and the microplastic concentrations were expressed as the number of microplastic particles per litre. All data were tested for normality using the Shapiro-Wilk test, which indicated that the data were normally distributed (P = 0.806). Additionally, the homogeneity of variance was tested using Levene's test, and the results confirmed that the data were homogeneous (P= 0.6074). Therefore, a one-way ANOVA was conducted, followed by a post-hoc Tukey's test, to analyse the differences between the wastewater treatment plants (significance was accepted at P < 0.05).

Removal efficiencies for each wastewater treatment plant were estimated by considering the concentration of microplastics in both influent and effluent (Fan et al., 2023; Franco et al., 2021) (Equation 3.1):

$$\text{Removal efficiency} = \frac{MP\ concentration\ influent - MP\ concentration\ effluent}{MP\ concentration\ influent} \times 100\% \qquad (3.1)$$

The amount of microplastics discharged per day by each wastewater treatment plant was calculated based on the average volume of effluent discharged (Murphy et al., 2016; Yang et al., 2019) (Equation 3.2):

$$\text{MP released} = \frac{MP\ present}{litres\ filtered} \times Average\ volume\ of\ effluent\ released(litres/day) \quad (3.2)$$

3.5 Limitations and Considerations

Despite rigorous QA/QC measures, certain limitations were acknowledged:

i. Temporal representation: Sampling occurred over a short period, providing only a snapshot of microplastic abundance in the rivers and the treatment performance of WWTPs. Seasonal variation in microplastic loads may influence their abundance in rivers and removal efficiencies in WWTPs.

ii. Sub-sampling for polymer identification: FTIR analysis was conducted on a limited number of particles due to time and cost constraints. While randomised, the sample may not capture the full diversity of microplastics.

iii. Potential degradation during digestion: Although wet peroxide oxidation was carefully controlled, prolonged exposure could potentially alter sensitive polymer structures.

These considerations were taken into account when interpreting the results, ensuring that conclusions regarding plant efficiency and microplastic release were conservative and robust.

CHAPTER FOUR

RESULTS AND DISCUSSION

4.1 Microplastics Concentration and Composition in Rivers

4.1.1 Abundance and Distribution of Microplastics in the Rivers

Microplastic particles were recorded at all the sampling sites, and the microplastic concentration ranged from 0.22 ± 0.026 to 0.917 ± 0.061 particles per litre (Figure 4.1). The overall mean microplastic concentration in the studied areas was determined to be 0.482 ± 0.037 particles per litre. Specifically, the concentrations were 0.562±0.035 particles per litre for the Nairobi River, 0.502±0.031 particles per litre for the Mathare River, and 0.371±0.042 particles per litre for the Ngong River. There was a significant difference in microplastic abundance among the three rivers (ANOVA: F (2, 42) =4.125; P= 0.0231). A post hoc Tukey's comparison revealed that the only statistically significant difference was observed between the concentration of microplastics in the Ngong River and the Nairobi River (P= 0.019). However, the differences in microplastic concentration between the Nairobi River and the Mathare River (P= 0.682) and between the Ngong River and the Mathare River (P= 0.196) were not statistically significant.

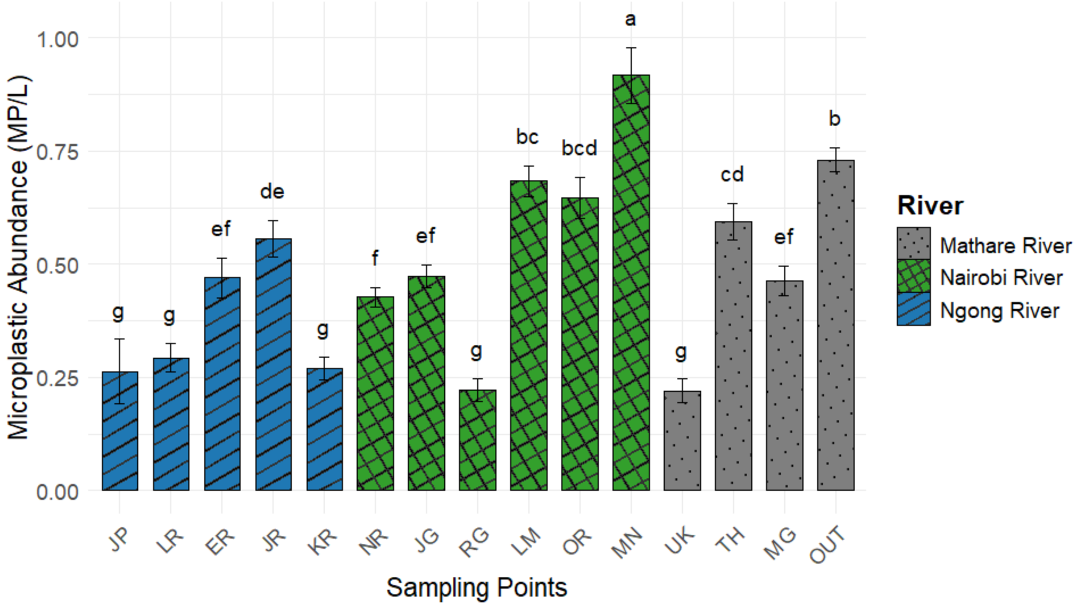

Figure 4.1: Microplastic Abundance in the Surface Waters of the Selected 15 Sites

The letters over the bars indicate significant differences ($p < 0.05$) in microplastic concentrations among sampling points. Bars that share the same letter indicate no significant differences, while different letters indicate significant differences between the sampling points. If letters overlap, it means there is no significant difference as long as they share at least one common letter. The error bars indicate the standard deviation.

The significant difference in microplastic concentration between the Nairobi River and the Ngong River can be attributed to the varying anthropogenic factors along the two rivers. The Nairobi River flows through intensely urbanised areas with high population pressure. It receives discharges from informal settlements, large city markets, and the wastewater treatment plants, which contribute substantially to the input of microplastics into the river (Bagnis et al., 2020; Mbui et al., 2016). Additionally, the differences in microplastic concentration between the rivers can be explained by variations in river flow and discharge (Skalska et al., 2020). The Mathare River, as a tributary, has relatively lower flow velocity and discharge, which likely reduces its capacity to transport microplastics, resulting in lower concentrations within its channel. In contrast, the Nairobi and Ngong Rivers have higher discharges and flow velocities, promoting the re-suspension of microplastics and leading to increased concentrations in their channels.

Significant differences ($P<0.05$) in microplastic abundance were observed across the different sampling points (Figure 4.1). The highest concentration of microplastics was observed along the Nairobi River at sampling point MN, which is located at the Junction of Mathare and Nairobi River (Figure 3.1). In contrast, the lowest concentration was found along the Mathare River at sampling point UK, located at Upper Kabete upstream (Figure 3.1). The high concentration of microplastics observed at 100m downstream of the confluence of Mathare and Nairobi River can be attributed to the confluence of the two rivers, which increases discharge and velocity. This leads to a rise in suspended particles as sediments from the riverbed are disturbed (Klein et al., 2015) . Additionally, the nearby poorly managed Dandora dumping site. This site serves as a primary disposal site for waste from the entire City of Nairobi (Bagnis et al., 2020). Without proper management, waste, including plastics, can directly reach the river system through leachate, surface runoff, and airborne pathways (Bagnis et al., 2020; Goonetilleke et al., 2020). In another study, Kazour et al. (2019) found that microplastic abundances are highest in sites near landfills, suggesting that environmental factors and microbial activities accelerate plastic degradation. Furthermore, the combined impact of the pollutants from both rivers may contribute to the increase in microplastic concentration. At this point, the two rivers are heavily polluted, carrying waste from the densely populated city centre and informal settlements. A study by Dahms et al.(2020) found a significant increase in microplastics after the confluence of two rivers. This finding further supports the assertion that tributaries play a role in the distribution of microplastics to larger rivers.

Microplastic concentrations are relatively low in the upstream sections of the Nairobi, Mathare, and Ngong rivers but increase downstream along the course of these rivers. The rise in concentration may be attributed to the greater discharge and higher water velocity as the rivers flow away from their sources, which enhances the transport of suspended microplastics. Additionally, changes in land use activities within the same drainage basin contribute to this increase. These rivers flow through the city centre, where industrialisation, informal settlements, and commercialisation have a significant impact. As a result, the rivers often serve as dumping sites for sewage and garbage, which can contribute to higher levels of microplastics. In another study in Spain, Gutiérrez-Rial et al. (2024) reported an increase in microplastic concentration from the upstream to the downstream sections of the Gafos River, Lagares River, and Miñor River in Pontevedra. This increase was attributed to human settlements and industrial activities, which were concentrated in the downstream sections of the river.

Interestingly, a notable decrease in microplastic concentration was observed at specific sampling points along the three rivers. For example, at the sampling point RG located in the Nairobi River at Westlands-Ring Road (Figure 4.1), there is a marked reduction. This decrease may be attributed to the presence of well-sewered residential areas upstream of this sampling point, where informal settlements are negligible, thus reducing the potential impact of wastewater discharges. A similar significant decrease in microplastic levels is noted at the sampling point KR at Kangundo Road (Figure 4.1). This point is situated further downstream of the Ngong River, and its low microplastic levels can likely be attributed to low flow and slow velocity, which may cause microplastics to settle along the river channel. Additionally, the anthropogenic activities at this point are relatively low compared to the midstream sections, leading to fewer potential sources of microplastics (Gutiérrez-Rial et al., 2024; Ngatia et al., 2023).

High concentrations of microplastics were observed at specific sampling points along the three rivers. For instance, at the sampling point OUT on the Mathare River (Figure 4.1). This sampling point is located downstream of the Mathare Slum, a large informal settlement. This area lacks adequate drainage and sewerage systems, resulting in raw and untreated sewage being discharged directly into the Mathare River. Similar high concentrations were noted at the sampling point (ER) at Enterprise Road and at the sampling point (JR) at Jogoo Road Bridge. Both of these sampling points are located downstream of the industrial areas, which may contribute to the elevated levels of microplastics. Additionally, a high concentration was noted at sampling point LM at Gikomba Market (Figure 4.1). This sampling point is located near the large city market, which contributes to the high concentration of microplastics. Studies by Duis & Coors (2016) and Horton & Dixon (2018) identified highly populated urban areas as significant sources of microplastic pollution. These areas serve as the centre for human activities that concentrate microplastic emissions (Rodríguez-Seijo & Pereira, 2017; Talbot & Chang, 2022). A study by Kunz et al. (2023) also reported a sudden increase in microplastic concentration from rural to urban areas along the main tributaries of the Wu River in Taicung, Taiwan. This sudden increase was attributed to the emissions from residential and commercial areas. Similarly, Mani et al. (2015) reported high microplastic levels in regions closer to densely populated areas along the Rhine River, a finding consistent with this study, which also found that sections draining populated areas had high microplastic concentrations.

The abundance of microplastics recorded in the three rivers was comparable to levels found in other urban catchments; however, the degree and intensity of urbanisation differ across these regions (Table 4.1). The concentration of microplastics was slightly higher than those observed in the Orange Vaal River, South Africa (Weideman et al., 2020) and the Upper St Lawrence River, Canada (Crew et al., 2020). In contrast, it was slightly lower than levels found in urban rivers in Shanghai Municipality, China (Wang et al., 2021). The higher levels in Shanghai were attributed to a few sampling sites receiving direct point-source pollution. These sites, therefore, had unusually high levels of microplastics. Overall, the findings from this study, consistent with those of other studies, indicate that microplastics are ubiquitous in river systems and are particularly abundant in densely populated areas. The observed concentrations suggest potential adverse ecological effects on aquatic organisms and may have implications for human health. Studies have reported the ingestion of microplastics by aquatic organisms and their transfer through the food chain (Bhuyan, 2022; Kosore et al., 2018). Given that these rivers serve as sources of potable water for domestic use, especially for the downstream community (Mbui et al., 2016; Ngatia et al., 2023), there are potential risks of human exposure.

Table 4.1: Comparison of Microplastic Abundance in Nairobi Rivers with Other Regions

Studied Area	Location	Abundance (particles/litre)	Reference
Nairobi River	Nairobi, Kenya (Urban)	0.562±0.035	This study
Mathare River	Nairobi, Kenya (Urban)	0.502±0.031	This study
Ngong River	Nairobi, Kenya(Urban)	0.371±0.042	This study
River Themi	Arusha, Tanzania (Urban)	0.22±0.026 - 0.917±0.061	(Kundu et al., 2022)
River Trent	Trent catchment, United Kingdom (UK), (Urban)	0.4	(Stanton et al., 2020)
Orange Vaal River	Johannesburg, South Africa (Urban/peri-urban)	0.23±0.27	(Weideman et al., 2020)
St Lawrence River	Montreal and Quebec City, Canada (Urban)	0.16±0.02	(Crew et al., 2020)
Urban rivers in Shanghai Municipality	Shanghai Municipality, China (Urban)	0.7 – 24.3	(Wang et al., 2021)

4.1.2 Microplastic Shape in the Rivers

The shape of microplastics is a crucial morphological property used to characterise their source, pathways, and origin, whether primary or secondary, as well as their potential to cause physical or physiological effects on exposed biota (Rodríguez-Seijo & Pereira, 2017). The recovered microplastics were sorted into six main shape categories: fibres, fragments, foam, granules, films, and pellets (Figure 4.2). The results showed that fibres (51.15%) were the most dominant microplastic type, followed by fragments (32.24%), and films (10.93%). Foam (4.89%), pellets (0.14%), and granules (0.65%) were less dominant, with pellets being present in 1 out of 15 sampling sites and granules in 4 out of 15 sampling sites under this study. Upon further analysis, it was found that the differences between the microplastic types were statistically significant ($P < 0.05$).

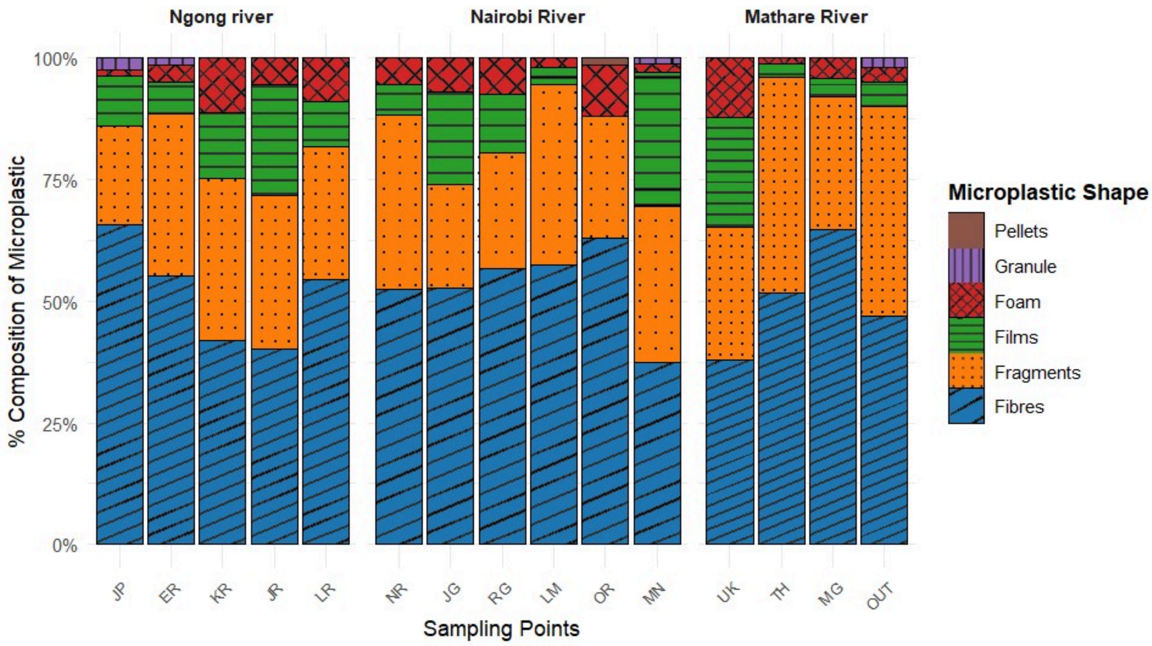

Figure 4.2: Percentage Composition of the Shape Distribution of Microplastic Recovered from the Different Sampling Points

The microplastic concentration of the different shapes did not vary significantly ($P>0.05$) between rivers, except for fibres. The concentration of fibre showed a significant difference between the Nairobi River and the Ngong River ($P= 0.008$). However, the differences in fibre concentration between the Nairobi River and the Mathare River ($P= 0.635$) and between the Ngong River and the Mathare River ($P= 0.132$) were not statistically significant.

The microplastic concentration for fibres, fragments, and films varied significantly ($p < 0.05$) between sampling points. The highest concentrations were observed to be in areas with high anthropogenic activities such as at Nairobi River at Outering (OR), located downstream of the city centre, Mathare River at Outering (OUT), downstream of the Mathare informal settlements, and the Nairobi River at the confluence of Nairobi and Mathare river (MN), which borders the poorly managed Dandora dumping site.

Fibres, as secondary microplastics, have numerous potential sources, including waste generated from textile industries, home laundering, and wastewater treatment plants (Liu et al., 2021; McGinnis, 2022). Additionally, fibres can originate from the breakdown of plastic particles such as plastic packaging materials, ropes, and food service items (Githaiga et al., 2023; McGinnis, 2022). The dominance of fibres in the Nairobi River and its tributaries may indicate a high level of domestic effluent, unregulated waste disposal practices, and the influence of runoff from urban activities (Githaiga et al., 2023). Synthetic fibres have been identified as a significant source of microplastic fibres from washing, with these fibres being found to dominate sewage disposal sites resulting from laundry activities (McGinnis, 2022). Browne et al. (2011) stated that thousands of textile fibres might be released into the aquatic environments after one washing. These fibres could either be sent to WWTP or directly released into aquatic environments. Since most sampled locations in the three rivers were adjacent to residential areas, especially informal settlements, domestic sewage could be a potential source of the fibres due to the poor drainage from houses that carry wash water.

Fragments, which were the second most dominant, have been associated with the breakdown and photo-degradation of larger plastic items such as food service products and packaging materials (Rodríguez-Seijo & Pereira, 2017). The numerous plastic products found carelessly discarded along the three rivers may contribute significantly to the formation of these fragments over time. Films are associated with the breakdown of larger plastic wraps and bags and are also used in agriculture to protect crops by suppressing weeds and conserving soil moisture (Duis & Coors, 2016; Tan, 2025). Plastic pellets, which are the least abundant, are secondary microplastics often occurring due to mishandling or direct waste discharges from plastic industries and are abundantly observed near industrial zones (Wang et al., 2021). Pellets are mostly reported to be abundant in sediments and rarely present in surface water columns, which explains their scarcity in the water samples from this study (Migwi et al., 2020).

The dominance of fibres reported in this study is consistent with findings from other studies. For instance, Scherer et al. (2020) reported that fibres were dominant, accounting for 46.5% of the total particle counts in the German River Elbe. The high proportion was attributed to the low settling velocity of fibres and the constant flow of water, which may prevent them from sinking. Similarly, Weideman et al. (2020) found that fibres are pervasive pollutants in the surface waters of the Orange-Vaal River, being dominant at even the most remote sampling sites. Kundu et al. (2022) also reported that fibres made up 40-72% of the microplastics recovered in the Themi River, Tanzania. Chen et al. (2020) also reported that fibres were the most dominant particle type in the Huangpu River watershed in China, accounting for 75%-95% of the total. The dominance of fibres in this study, consistent with the findings from other studies, highlights their widespread presence in freshwater ecosystems. This prevalence is linked to the effluent discharges from household sewage, which often includes the washing of synthetic textiles, as well as discharges from wastewater treatment plants and the breakdown of plastic items.

4.1.3 Microplastic Colour in the Rivers

Colour is of particular concern in aquatic environments, as most coloured microplastic particles are often misinterpreted by aquatic organisms as food particles (He et al., 2020). Additionally, colour is expected to provide information about microplastic sources. Microplastics were categorised into various colour classes, including white, transparent, red, yellow, brown, green, and black, as shown in Figure 4.3. White microplastics (40.08%) were the most abundant particle, followed by brown (27.97%), and then transparent (10.52%). The least abundant were red (3.79%) and green (0.73%). The differences in microplastic colours were statistically significant ($P < 0.05$).

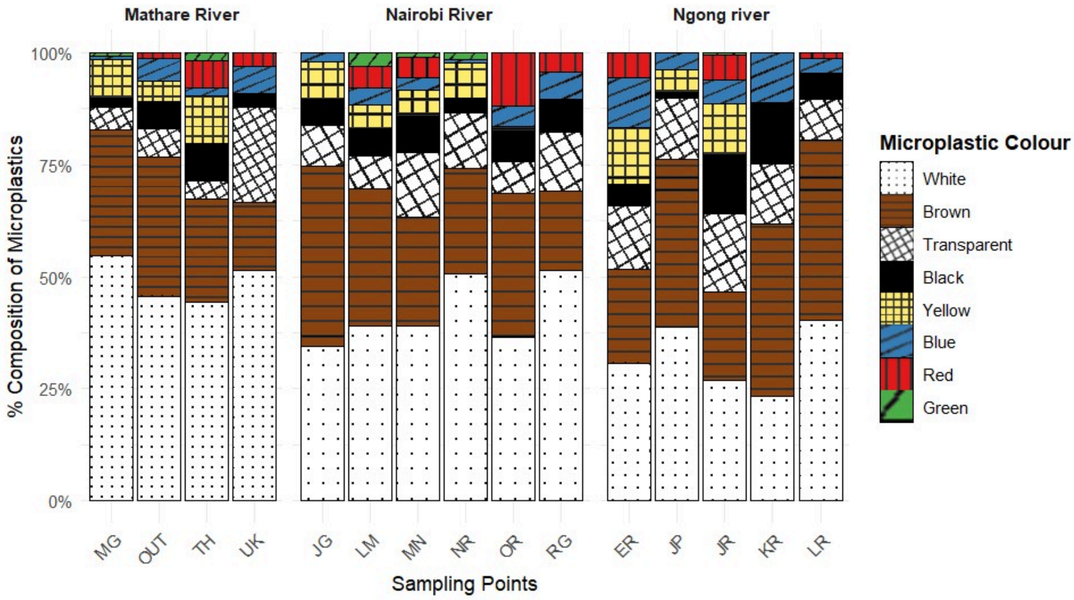

Figure 4.3: Percentage Composition of Colour Distribution of Microplastic Recovered from the Different Sampling Points

The microplastic concentration of the different colours did not vary significantly (P>0.05) between the rivers, except for white and brown particles. For white microplastics, there was a significant difference in concentration between the Ngong River and Nairobi River (P= 0.0004) and between the Ngong River and Mathare River (P= 0.0003). However, there was no significant difference between the Mathare River and Nairobi River (P= 0.85). For brown microplastics, a significant difference was observed between the Ngong River and the Nairobi River (P= 0.03). However, there was no significant difference between the Ngong River and Mathare River (P= 0.52) or between the Nairobi River and Mathare River (p= 0.40). The microplastic concentration of transparent, yellow, white, and brown particles varied significantly (P<0.05) between sampling points. However, the microplastic concentration for the other colours did not vary significantly (P>0.05).

The observed prevalence of white and transparent microplastics in this study may be associated with the loss of their original colour due to photo-degradation (Wagner & Lambert, 2018). White particles are also associated with polyethylene (PE), a polymer commonly found in consumer products used in food packaging and household items in urban areas (Rodríguez-Seijo & Pereira, 2017). Some of the products include water bottles, plastic packaging bags, single-use plastics, food containers, disposable plates, forks, and spoons (Osorio et al., 2021). Their prevalence may also be attributed to their high visibility against darker backgrounds, making them easier to detect during sampling and leading to higher detection rates (Hidalgo-Ruz et al., 2012). However, using spectroscopic confirmation can help reduce bias.

Other colours, such as red, yellow, brown, green, and black, are possibly derived from commonly used products, including household items, packaging bags, food wrappers, toys, and ropes (Hidalgo-Ruz et al., 2012; Osorio et al., 2021). Black colour is associated with black plastic products, such as mulch films and plastic bags. It is more commonly linked to waste collection bags, household plastic bags, packaging materials, and refuse liners frequently used in the study areas (Zhang et al., 2024). Yellow particles have been associated with aged and photo-oxidised plastics, which tend to discolour with UV exposure and environmental weathering (Hidalgo-Ruz et al., 2012). Coloured microplastics are commonly used in consumer goods to enhance market appeal (Migwi et al., 2020), suggesting that many microplastics originate from the breakdown of large coloured plastic items, such as clothing and packaging, found in surrounding areas (Duis & Coors, 2016). Microplastics can inherit their colours from their parent plastic products, but may change colour due to photo-degradation and their residence time in the water (Hidalgo-Ruz et al., 2012; Wagner & Lambert, 2018).

The predominance of white microplastic particles has been reported in previous studies. For instance, Kerubo et al. (2020) reported that 69.6% of the recovered microplastics were white particles along the Kenyan coast. Similarly, Kosore et al. (2018) also recorded 51% as white particles on the Kenyan coastline. Additionally, Goonetilleke et al. (2020) noted that white microplastics were the dominant colour in the Brisbane River, Australia. These findings highlight the prevalence of white microplastic particles, as observed in this study and across other studies. However, the colour derived from the original plastic can change due to degradation, making it challenging to identify the sources of the microplastic. Colour is critical because aquatic organisms may mistakenly ingest microplastics due to their resemblance to their prey. For instance, Zhang et al. (2020) found blue microplastics to be the most common colour observed in the digestive tracts of sampled fish, with other colours including black and white. Kosore et al. (2018) also found that black microplastics were the most commonly ingested microplastic colour by zooplanktons. Therefore, the presence of coloured microplastics observed in this study is a significant concern as they pose potential ecological and human health risks.

4.1.4 Microplastic Polymer in the Rivers

The FTIR spectroscopy identified all analysed particles, which were categorised into six different types. The polymers identified include Polyethylene (PE), Polystyrene (PS), Polyethylene Terephthalate (PET), Polyester (PES), Polyurethanes (PUR), and Cellulose acetate. PE (43.33%) was the most dominant, followed by PS (18.33%), then PET (11.67%), and the least abundant polymer was cellulose acetate (6.67%), as shown in Figure 4.4.

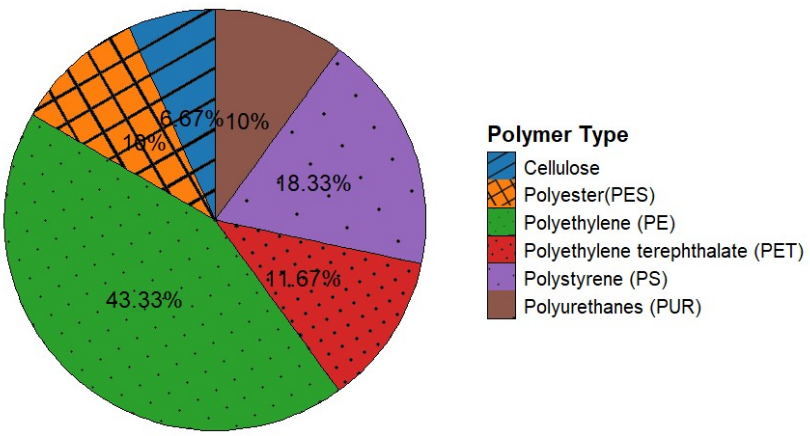

Figure 4.4: Percentage Proportion of Microplastic Polymers from the Studied Area

PE and PS, the most abundant, are widely used in consumer-related products. PE is primarily associated with packaging films, pipes, plastic bags, agricultural films, household utensils, and cookware with plastic coatings (Goonetilleke et al., 2020; Rodríguez-Seijo & Pereira, 2017). Non-stick cookware made from PE and Polytetrafluoroethylene (PTFE) has been identified as a source of microplastics, due to high temperature, scratches, cracks, and tears in the surface coating (Cole et al., 2024). PE has also been associated with white particles (Rodríguez-Seijo & Pereira, 2017), which were the most abundant in this study. PS, on the other hand, is commonly used in single-use items such as disposable plates, cutlery, cups, and food packaging, especially in foam, insulation materials, and protective packaging. Numerous disposable products, including water, milk, and juice containers, cutlery, toys, bags, and household items, are used in the study area and were observed to be indiscriminately disposed of along the rivers. These consumer products are possible sources of these polymers in the three rivers.

The predominance of PE and PS in surface waters can also be attributed to their density relative to water, as their density is lower than that of water (Razeghi et al., 2021). High-density polymers such as PET, commonly found in clothing and water bottles (Rodríguez-Seijo & Pereira, 2017), were detected in surface water. This presence may be attributed to resuspension, which is influenced by flow velocities and turbulence. Such findings have been noted in surface waters and reported in other studies (Chen et al., 2020; Migwi et al., 2020). Cellulose acetate, which is the least observed, is a chemically modified biopolymer considered a semi-synthetic polymer, primarily used in plastic films, textiles, photographic films, and cigarette filters (Wagner & Lambert, 2018; Weideman et al., 2020). Although derived from natural cellulose, cellulose acetate is chemically modified and exhibits persistence in aquatic environments, thereby posing a risk of ingestion by aquatic organisms and leaching associated additives (Mušič et al., 2023; Yadav & Hakkarainen, 2021). Their persistence in the environment is greater than that of natural fibres but slower than that of synthetic polymers (Yadav & Hakkarainen, 2021).

Polymer identification may be influenced by environmental processes, such as fragmentation, photo-oxidation, and mechanical weathering, which can alter their physical and chemical properties over time (Wagner & Lambert, 2018). Such degradation can sometimes complicate polymer identification as aged or weathered plastics may show inconsistent results or reduced signal clarity (Fernández-González et al., 2021). Moreover, while ATR-FTIR spectroscopy is suitable for polymer characterisation, it has limitations in detecting heavily weathered particles or very tiny microplastics (<500 µm) (Razeghi et al., 2021; Wagner & Lambert, 2018).

The predominance of PE polymers has been reported in previous studies. In the upper River Tame in the West Midlands, UK, Tibbetts et al. (2018) found PE as the most dominant polymer, accounting for 50% of the recovered microplastic. This dominance was attributed to the significant use of PE in manufacturing plastic packaging and containers. Similarly, Scherer et al. (2020) reported that PE had the highest concentration in the European river, accounting for 47.5% of the identified microplastics, which was attributed to its high production. Furthermore, Zhang et al. (2020) found PE as the most abundant polymer in the surface water of the Ma'an Archipelago in China, accounting for 30% of the microplastics recovered. This prevalence was attributed to the everyday use of fishing gear and also to their lower density. Liu et al. (2021) also found PE to be the chief component of the polymers identified in the Haihe River, China, accounting for 49.3%. This finding highlights the widespread use of PE, which is commonly sourced from consumer-related products.

4.2 Relationship Between Microplastics Abundance and Anthropogenic Activities

The land uses within each sub-catchment were categorised into seven distinct types, including built-up areas, forests, water bodies, agriculture, plantations, bare lands, and wetlands (Figure 4.5). Agricultural land and built-up areas were the most dominant land use types in most catchments, while wetlands and water bodies had the least coverage (Table 4.2).

When determining the correlation between microplastic abundance in surface water and different land use types, the Pearson Product-Moment correlation coefficients indicated a significant positive correlation between microplastic abundance and built-up areas ($r = 0.62$, $P = 0.013$), and a significant negative correlation with forest ($r = -0.54$, $P = 0.037$) and wetlands ($r = -0.53$, $P = 0.041$). These land uses were significantly associated with an increase or decrease in microplastic abundance. The other land use types did not show a significant relationship. Specifically, plantations, water bodies, and bare lands showed a negative correlation with microplastic abundance, whereas agriculture showed a positive relationship; however, these findings were not statistically significant (Table 4.3).

When examining the relationship between land use and the different shapes of microplastics, a significant positive correlation was found with built-up areas for both fibres (r= 0.61, P= 0.015) and fragments (r= 0.65, P= 0.008). However, the other shapes of microplastics did not show any significant relationship with land use, suggesting that anthropogenic activities may not influence their abundance in the rivers. The statistical significance of these findings is shown in Table 4.3.

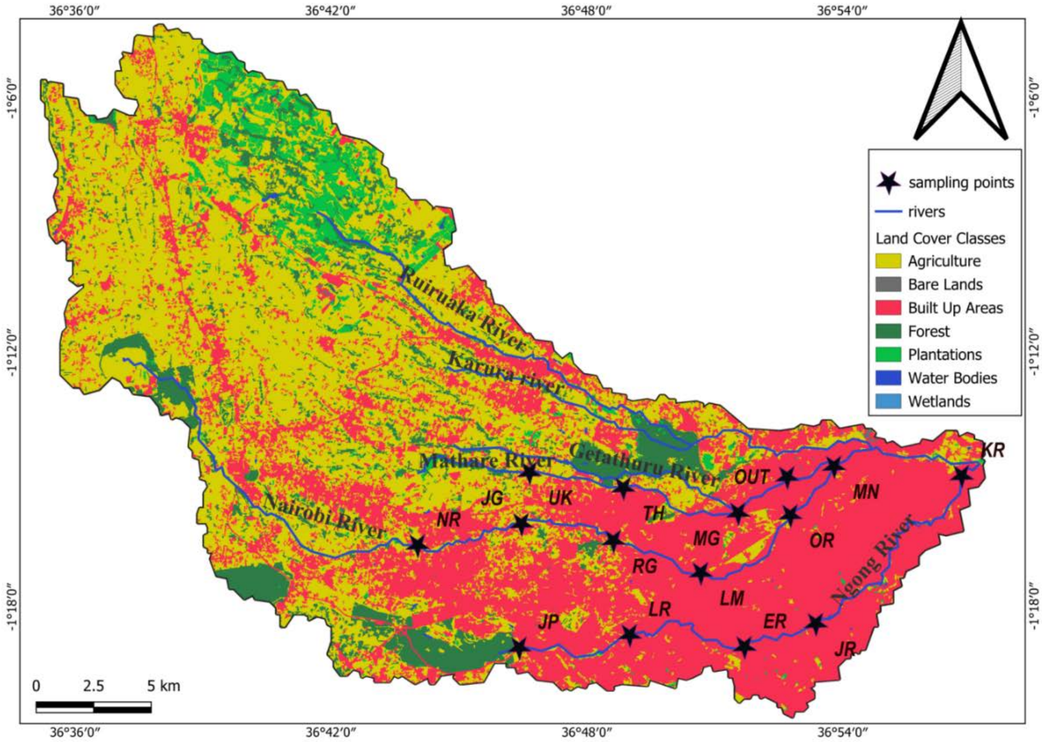

Figure 4.5: Land Use of the Study Area

Table 4.2: Land Use Data for Each Sub-catchment in the Nairobi River Basin

Sampling point	Catchment Areas (Km²)	Proportion of catchment areas						
		Agriculture	Bare Lands	Built-Up	Forest	Plantation	Water bodies	Wetlands
JP	30.34	0.3082	0.0004	0.3039	0.2604	0.1250	0.0018	3.63E-05
LR	45.50	0.3059	0.0006	0.3438	0.2548	0.0934	0.0012	2.42E-05
ER	70.43	0.3870	0.0005	0.3813	0.1589	0.0712	0.0009	1.56E-05
JR	86.58	0.4538	0.0008	0.3704	0.1155	0.0585	0.0007	1.27E-05
KR	103.63	0.4005	0.0007	0.4268	0.1212	0.0491	0.0013	1.06E-05
NR	104.86	0.4245	5.72E-05	0.4040	0.1076	0.0631	0.0005	0
JG	115.47	0.4235	8.14E-05	0.4076	0.0985	0.0596	0.0005	0
RG	143.59	0.4167	8.71E-05	0.4241	0.0897	0.0686	0.0006	0
LM	150.51	0.4087	8.30E-05	0.4369	0.0863	0.0672	0.0005	0
OR	164.38	0.4016	7.60E-05	0.4540	0.0791	0.0644	0.0005	0
MN	293.00	0.4096	0.000116	0.4358	0.0864	0.0672	0.0007	0
UK	31.75	0.4035	0	0.4068	0.0828	0.1060	0.0006	0
TH	50.65	0.4045	0	0.4316	0.0673	0.0953	0.0011	0
MG	118.96	0.3713	0	0.4147	0.1031	0.0752	0.0011	0
OUT	123.68	0.4056	4.29E-05	0.4541	0.0996	0.0735	0.0011	0

Table 4.3: Pearson Product Moment Correlation Coefficient between Microplastic Abundance and Land Use Data

Land use	Agriculture	Bare land	Built Up	Forest	Plantations	Water bodies	Wetlands
Microplastic Abundance	0.47	-0.39	0.62*	-0.54*	-0.41	-0.41	-0.53*
Fibres	0.11	-0.44	0.61*	-0.37	-0.31	-0.14	-0.41
Fragments	0.28	-0.33	0.65*	-0.45	-0.29	-0.07	-0.44

***Significant correlation (p<0.05).**

Microplastics in the environment are primarily of anthropogenic origin, influenced mainly by human activities (Su et al., 2020; Yonkos et al., 2014). A positive correlation between microplastic concentration and built-up areas indicates that higher microplastic concentrations were found in these areas, including industrial zones, residential areas, and commercial centres. Built-up areas are notable hotspots for microplastics, with these pollutants entering the river system from various sources, including industrial discharges, road runoff, wastewater treatment effluent, domestic wastewater, and indiscriminate disposal of plastic waste (Horton & Dixon, 2018; Townsend et al., 2019; Zhang et al., 2024). The three rivers receive waste from the densely populated and industrialised city centre. This waste originates from informal settlements, poorly managed dumping sites, industrial discharges, sewer overflows, and commercial centres, all of which contribute significant amounts of microplastics at sampling points within the city centre. The correlation between fibres and fragments with built-up areas suggests that their sources may be linked to discharges from these areas. Numerous studies have confirmed the positive correlation between microplastic quantities and built-up areas or densely populated regions. For instance, Kunz et al. (2023) reported a strong positive correlation between microplastic concentration and areas of industrial and commercial development in the Toubiankeng River, Taiwan. Similarly, Su et al. (2020) found that microplastic pollution in the Great Melbourne Area, Australia, was closely associated with transportation, industrial, and commercial land uses. Additionally, Yonkos et al. (2014) identified a positive correlation between microplastic pollution and the percentage of built-up areas and industrial areas in estuarine rivers of the Chesapeake Bay in the USA.

Microplastic abundance was noted to have a negative relationship with sites located further from anthropogenic activities, particularly in forested regions. These areas tend to have less intensive human activities, resulting in fewer potential sources of microplastics. Forested land is associated with lower population density, less development, less impermeable surface and fewer drainage systems (Yonkos et al., 2014). Kunz et al. (2023) reported a strong negative correlation between microplastics and forested areas in the Toubiankeng River, Taiwan. Similarly, Townsend et al. (2019) found a negative correlation between microplastics and less built-up areas in the Greater Melbourne Region, Victoria, Australia. Although not statistically significant, Yonkos et al. (2014) also observed a negative relationship between microplastics and forested land use.

A significant negative correlation was also observed between microplastic abundance and wetlands. Wetlands are considered important sinks for many pollutants, as the vegetation in these areas acts as an effective retaining medium (Li et al., 2018). The water flow in the wetlands is slow, allowing it to be retained for a period before flowing into the receiving ecosystem. As a result, pollutants in the rivers are intercepted by the vegetation in the wetlands, which helps to reduce the presence of microplastics as they become trapped in the wetland (Townsend et al., 2019). This phenomenon may explain the lower concentration of microplastics than expected at sampling point LR along the Ngong River. This point is located downstream of the Kibera slum, the largest informal settlement in Kenya, which likely contributes to an increased concentration of microplastics from the wastewater discharged into the river. However, the river flows into the Nairobi dam, recognised as an urban wetland (Ngatia et al., 2023), before being sampled at point LR. The presence of this urban wetland likely contributes to a significant reduction in the concentration of microplastics transported from the informal settlement.

Although other land use types, such as plantations, agriculture, water bodies, and bare lands, did not show a statistically significant correlation with microplastic abundance, the direction of their relationship suggests an influence. The negative correlation observed for bare lands, plantations, and water bodies indicates that these areas are associated with reduced microplastic inputs. This may be due to lower human activities and natural buffering processes that decrease runoff in the case of plantations. In contrast, the positive correlation observed for agricultural land, although not statistically significant, may reflect the potential impact of farming activities that involve the use of plastic mulch, greenhouses, irrigation pipes, and fertiliser bags (Grbić et al., 2020). However, the lack of statistical significance suggests that these activities are not strong enough to account for variations in microplastic levels within these land uses. This implies that these land uses may contribute relatively fewer direct sources of microplastics and that the pathways through which microplastics enter rivers are likely more diffuse (Talbot & Chang, 2022). Furthermore, other predominant sources, such as built-up areas that include industrial sites, commercial areas, and densely populated regions, play a significant role in influencing the distribution of microplastics.

The land use analysis indicates that anthropogenic activities significantly contribute to microplastic contamination, with higher concentrations found in built-up areas and lower concentrations in undeveloped regions, such as the forested areas and wetlands. While this study focused on land use as a driver of microplastic distribution, other environmental and hydrological factors, such as river flow and rainfall intensity, may also influence microplastic distribution and interact with the land use effects. However, these factors were not explored in this study. Future research could enhance the understanding of the factors influencing microplastic abundance by incorporating these elements, thus providing a more comprehensive insight into the mechanisms driving microplastic distribution in river systems.

4.3 Efficacy of Selected Wastewater Treatment Plants (WWTPs) in Removal of Microplastics

4.3.1 Microplastic Abundance in the Wastewater Treatment Plants

Microplastics were found in all collected samples, with varying levels of abundance at both facilities (Figure 4.6). The concentration in the Ruai wastewater treatment plant was 9.99 ± 0.50 particles per litre in the influent and 1.08 ± 0.18 particles per litre in the effluent. The Kariobangi wastewater treatment plant had a lower concentration, with 4.70 ± 0.28 particles per litre in the influent and 1.03 ± 0.08 particles per litre in the effluent. An analysis of Variance (ANOVA) indicated significant difference between the influents of the two wastewater treatment plants ($P<0.0001$), and between the influent and effluent of each plant ($P<0.0001$). However, no significant difference was observed between the effluents of the two plants ($P= 0.9967$).

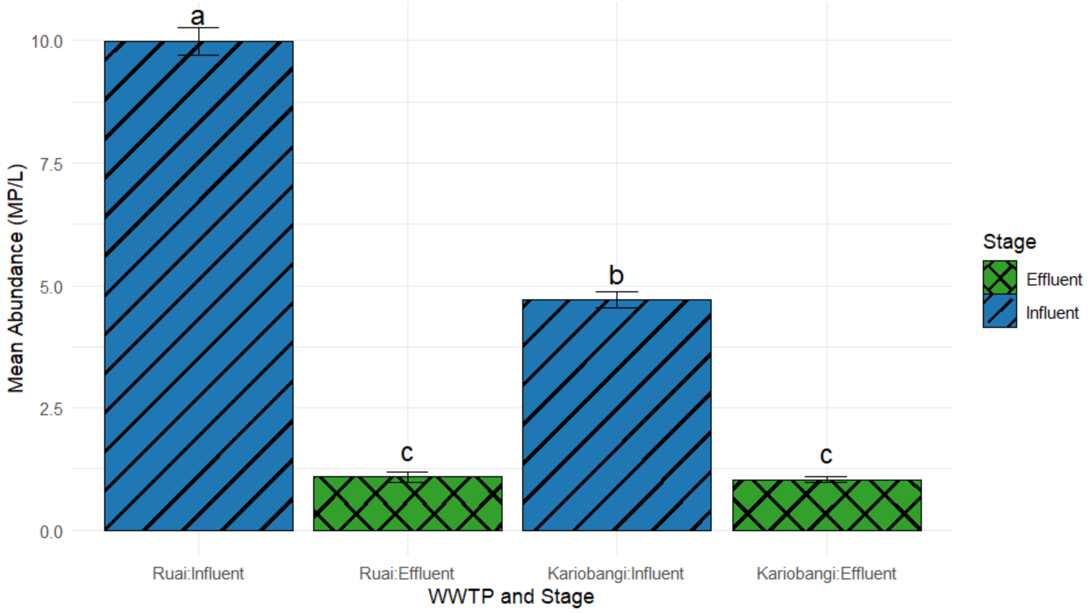

Figure 4.6: Abundance of Microplastics (particles/litre) in the Influent and Effluent of the two Wastewater Treatment Plants

The letters above the bars indicate significant differences ($p < 0.05$) in microplastic concentration between the wastewater treatment plants and stages. Bars sharing similar letters show no significant differences, while different letters signify significant differences. The error bars represent the standard deviation.

The significant difference in microplastic abundance between the influent of the two wastewater treatment plants can be attributed to factors such as the size of the population served, the wastewater source, and the surrounding land use (Franco et al., 2021; Sun et al., 2019). Both wastewater treatment plants are located within the urban areas of Nairobi, but they have different hydraulic capacities. The Ruai wastewater treatment plant serves a larger population and receives wastewater from both domestic and industrial sources, which may contribute to the higher concentrations of microplastics observed. In contrast, the Kariobangi wastewater treatment plant primarily treats domestic wastewater and serves a smaller population, which results in significantly lower concentration of microplastics in its influent.

The concentration of microplastics in the effluents of both wastewater treatment plants significantly decreased compared to the influent ($P<0.0001$). The differences in microplastic concentration in the final effluent can primarily be attributed to the wastewater treatment processes and technologies used (Conley et al., 2019; Cristaldi et al., 2020). Both treatment plants employ secondary treatment, but they utilise different treatment technologies. For instance, Kariobangi WWTP utilises primary clarifiers that have a hydraulic holding time of approximately 2 hours. This design promotes settling of solids before the biological filtration process. In each primary clarifier, floating solids are removed by skimming the water surface, which primarily removes floating microplastics at this stage. Heavier microplastics, along with those embedded in solid flocs, can be removed during the grit removal process and in primary clarifiers through settling and gravity separation (Conley et al., 2019; Sun et al., 2019). The biological filters may further reduce the concentration of microplastics, as sludge flocs can facilitate the accumulation of the plastics, which could then be settled in the secondary clarifiers (Liu et al., 2020; Sun et al., 2019). Several studies have shown that most microplastics are removed during the primary treatment process (Carr et al., 2016; Gies et al., 2018; Lares et al., 2018; Mintenig et al., 2017). In a pilot study, Liu et al. (2020) found that biofilters could reduce microplastics by 79-89% in a wastewater treatment plant in Hvidovre, Denmark. Additionally, Gies et al. (2018) reported a 91.7% decrease in microplastics after primary clarification in a wastewater treatment plant in Vancouver, Canada.

The removal of microplastics at the Ruai wastewater treatment plant can be attributed to the extended retention time of wastewater within the lagoon system, which exceeds two months. These include 10 days for anaerobic ponds, 35 days for facultative lagoons, and 15 days for maturation lagoons. The prolonged contact time enhances the interaction between microplastics and wastewater throughout the treatment processes. This results in a more significant settling effect, promoting other activities such as adsorption, biological uptake of microplastics, and aggregation (Fan et al., 2023; Sun et al., 2019). Carr et al. (2016) noted that longer contact times are associated with an increased likelihood of surface biofilm formation on microplastics, which increases their relative density. Microplastics can be adsorbed or captured by sludge flocs and eventually be removed through gravity settling or ingested by protozoa or metazoan

(Fan et al., 2023). Studies have reported that significant removal of microplastics is achieved through the settling of microplastics (Fan et al., 2023; Habib et al., 2020; Sol et al., 2020).

Although wastewater treatment plants can partially remove microplastics through processes such as skimming and sedimentation, these methods are not specifically designed to eliminate them. As a result, a considerable amount of microplastics may still be discharged with the effluent and enter the receiving water systems. Furthermore, some microplastics can accumulate in sewage sludge, which may release additional microplastics into the environment when the sludge is applied to land (Cristaldi et al., 2020; Habib et al., 2020; Magni et al., 2019). Therefore, the management of sludge plays a crucial role in determining the fate of the embedded microplastics.

The influent and effluent abundances observed in this study are consistent with findings from other studies, although most of these studies reported lower concentrations in the effluent. For instance, Murphy et al. (2016) conducted a study at a secondary wastewater treatment plant in Glasgow, Scotland, where they found the number of microplastics decreased from 15.70 ± 5.23 microplastics per litre in the influent to 0.25 ± 0.04 microplastics per litre in the effluent. Similarly, Yang et al. (2019) reported that the concentration of microplastics in the influent water in Beijing, China, was 12.03 ± 1.29 items per litre, while the final influent contained 0.59 ± 0.22 items per litre. Additionally, Long et al. (2019) found that microplastics decreased from 6.55 particles per litre in the influent to 0.59 particles per litre in the effluent in Xiamen, China. The differences between these studies can be mainly due to the treatment technologies used, the size of the service population, operational conditions, and methodological approaches to sampling and analysis (Conley et al., 2019; Lares et al., 2018).

4.3.2 Microplastic Shape in the Wastewater Treatment Plants

Microplastic shape is an important factor in characterising microplastics, as it not only influences their removal efficiency in wastewater treatment plants but also impacts their interactions with other contaminants and microorganisms in wastewater (Franco et al., 2021; Sun et al., 2019). Microplastics were categorised into five distinct shapes: fibres, fragments, film, pellets, and foam. Fibres accounted for the highest proportion of observed microplastics in both influent and effluent, with average percentages of 60.8% and 70.9% in the influent and 53.8% and 47.3% in the effluent of Ruai and Kariobangi wastewater treatment plants, respectively. Fragments were the second most dominant shape, followed by films, foam, and pellets, which were less common, each with an average below 10% (Figure 4.7). The distribution of microplastic shapes in the effluent resembled that in the influents, except for pellets in both plants and foam in the Kariobangi wastewater treatment plant, which were removed. The results presented in Figure 4.7 display the percentage composition of each microplastic type in the influent and effluent of each WWTP, rather than their absolute abundance.

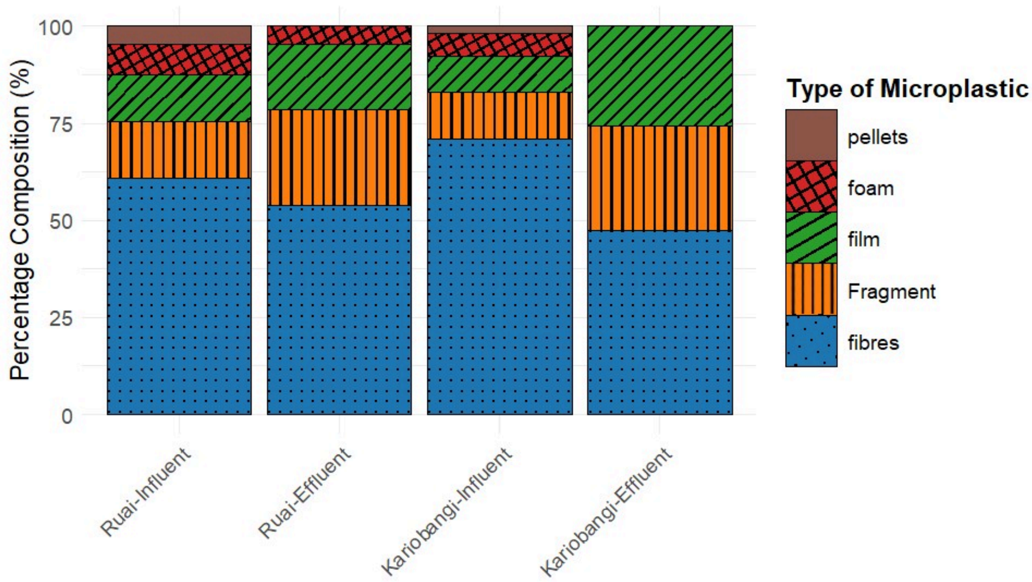

Figure 4.7: Percentage Composition of Different Microplastic Shapes in the Influent and Effluent of the Ruai and Kariobangi Wastewater Treatment Plants

The presence of numerous fibres can be attributed to the release of fibres during the laundry practices (Browne et al., 2011; Napper & Thompson, 2016; Sol et al., 2020). Browne et al. (2011) reported that a single wash can release more than 1900 textile fibres into the aquatic environment. Napper & Thompson (2016) also indicated that laundering 6 kg of synthetic materials in a domestic washing machine can discharge up to 700,000 fibres into the sewage system per wash. The Ruai and Kariobangi wastewater treatment plants primarily serve the densely populated urban areas of Nairobi, which include both informal and formal settlements. These settlements can generate substantial quantities of fibres through domestic laundry, making them a significant source of fibrous microplastics. Additionally, the textile manufacturing industries can contribute considerable amounts of microplastics (Lares et al., 2018) that are released through the industrial sewerage system and subsequently enter the wastewater treatment plants.

The abundance of fibres in the effluent can also be attributed to their morphological characteristics, which make them difficult to remove during treatment. Fibres are long and narrow, which inhibits their retention in conventional wastewater treatment plants (Cristaldi et al., 2020; Sol et al., 2020; Sun et al., 2019). The significant presence of fibres in the effluent from both wastewater treatment plants may be a primary source of this type of microplastic in the receiving water bodies. Studies have found fibres to be the dominant type of microplastic in river ecosystems (He et al., 2020; Kundu et al., 2022; Wang et al., 2021), indicating that wastewater treatment plants are one of the sources.

The fragments likely originate from commonly used consumer products such as plastic bottles, toys, containers, and materials used for decorating and construction. These items are often disposed of indiscriminately in the environment. They can break down through various processes, including chemical, biological, or photo-degradation, into small pieces as secondary microplastics, characterised by their irregular shapes (Franco et al., 2021; Hidalgo-Ruz et al., 2012; Liu et al., 2019). Fragments have been observed to be the second most abundant type of microplastics after fibres, as noted in other studies (Blair et al., 2019; Magni et al., 2019; Mason et al., 2016).

Film and foam, which are less abundant, are mainly derived from the fragmentation of plastic bags used for packing products and wraps (Rodríguez-Seijo & Pereira, 2017). Pellets, on the other hand, are primarily composed of primary microplastics used in personal care items (Sun et al., 2019). Other studies have reported that foam and pellets are significantly less abundant components of microplastics found in WWTPs (Franco et al., 2021; Habib et al., 2020; Talvitie et al., 2017). Pellets were observed to be removed during the treatment process, suggesting that they were more likely to be captured by these systems. Similarly, foam was noted to be removed in the Kariobangi wastewater treatment plant. Previous studies have found that the shapes of microplastics can influence their removal efficiency (Murphy et al., 2016; Talvitie et al., 2017). In this study, only pellets and foam demonstrate preferential removal. However, other studies have also found that shape does not significantly influence removal (Wang et al., 2020).

The results of this study confirm that fibres are the predominant microplastic shape, aligning with findings from other studies. Akarsu et al. (2019) reported that fibres accounted for 69.7-79% of microplastics in influent and 32.8% in effluent at the wastewater treatment plants in Mersin Bay, Turkey. Gies et al. (2018) identified fibres as the most dominant microplastic shape in a wastewater treatment plant in Vancouver, Canada, comprising 65.6% of the microplastics. Similarly, Blair et al. (2019) reported that fibres accounted for 67% of all microplastic items found in a wastewater treatment plant in Scotland, UK. Collectively, these studies indicate that fibres are the dominant type of microplastic particles present in wastewater, with variations depending on the source and type of wastewater treatment plant.

4.3.3 Microplastic Colour in the Wastewater Treatment Plants

The colour of microplastics is crucial due to their potential adverse effect on aquatic species. Coloured microplastics are more likely to be ingested by these species because they can be mistaken for their natural prey (Bayo et al., 2020; Chen et al., 2020; Wang et al., 2020). In the samples analysed, eight distinct colours were identified: transparent, black, white, blue, red, green, yellow, and brown. Among these, transparent was the most dominant colour in both wastewater treatment plants at both the influent and effluent stages. It accounted for 28.69% and 35% of the influent and effluent samples at the Ruai WWTP, and 26.06% and 39.78% of the influent and effluent samples at the Kariobangi WWTP. The results illustrated in Figure 4.8 show the percentage composition of each microplastic colour in the influent and effluent of each WWTP, rather than their absolute abundance.

The profiles of microplastic colours in the influent samples were similar for both wastewater treatment plants, except for yellow particles, which were only found in the Ruai samples. At the effluent stage, brown particles were removed from the Kariobangi wastewater treatment plant, while yellow particles were removed from the Ruai wastewater treatment plant (Figure 4.8).

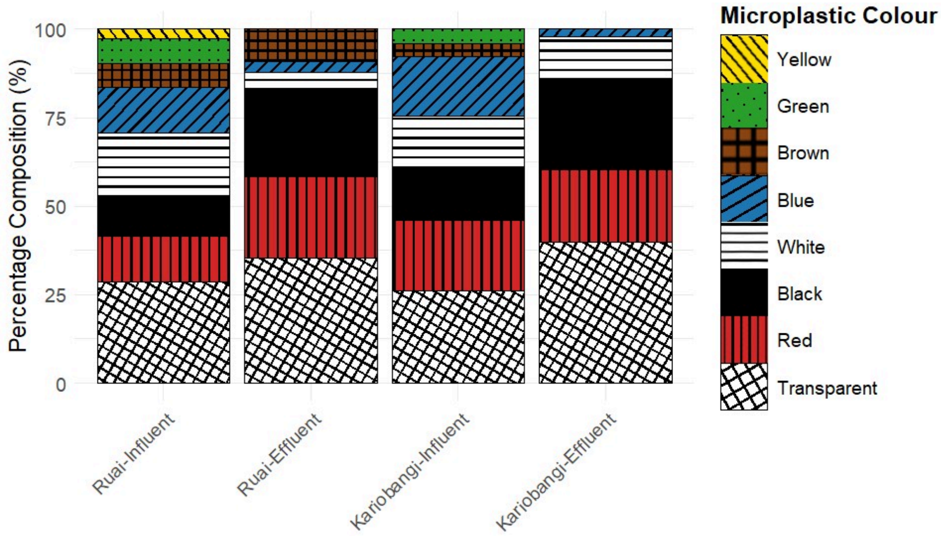

Figure 4.8: Percentage Composition of Different Microplastic Colours in the Influent and Effluent of the Ruai and Kariobangi Wastewater Treatment Plants

The presence of diverse colours in plastics often indicates the use of different dye types during production and suggests varied origins such as consumer products, packaging materials, and textiles, which are frequently coloured to enhance market appeal (Bayo et al., 2020; Duis & Coors, 2016). However, the colour of plastics can change due to factors such as ultraviolet oxidation and their duration in water (Wang et al., 2020).

In Nairobi, residents use transparent packaging bags and containers in their daily lives, contributing to the widespread presence of transparent microplastics. Transparent particles being dominant aligns with findings from Mintenig et al. (2017), who reported that transparent microplastics accounted for 59.2% of the total microplastics in Saxony, Germany. Additionally, Wang et al. (2020) identified transparent microplastics as the most abundant in both the influent and effluent of the wastewater treatment plant in Changzhou, China. Furthermore, Liu et al. (2019) revealed that transparent particles were the dominant colour in Wuhan City, China, accounting for 56%. These results indicate the abundance of transparent microplastics in most wastewater treatment plants, suggesting that consumer products may be a potential source. However, the abrasion and breakdown of microplastics can influence the colour of microplastics.

4.3.4 Microplastic Polymer in the Wastewater Treatment Plants

A subsample of microplastics was analysed from both the influent and effluent samples for polymer identification. In total, six different polymer types were identified using FTIR Spectroscopy: polyethylene terephthalate (PET), polypropylene (PP), polyethylene (PE), polystyrene (PS), Low-density Polyethylene (LDPE), High-Density Polyethylene (HDPE), and Polyester (PES) (Figure 4.9). Among these, LDPE was found to be more prevalent in Ruai's influent and effluent, accounting for 38.5% and 27.3%, respectively. Conversely, PP was more dominant in the influent and effluent of the Kariobangi WWTP, accounting for 33.3% and 30.8%, respectively. HDPE and PES were less common in both WWTPs, each constituting less than 10%. The results presented in Figure 4.9 illustrate the microplastic polymer data as percentage composition. The differences observed between influent and effluent indicate changes in the relative proportions of polymers rather than absolute changes in the types of polymer present.

A comparison of the polymer types between the influent and effluent indicated that the distribution of microplastic polymers in the effluent resembled that of the influent. However, fewer polymer types were present in the effluent. Notably, HDPE was entirely removed in both wastewater treatment plants.

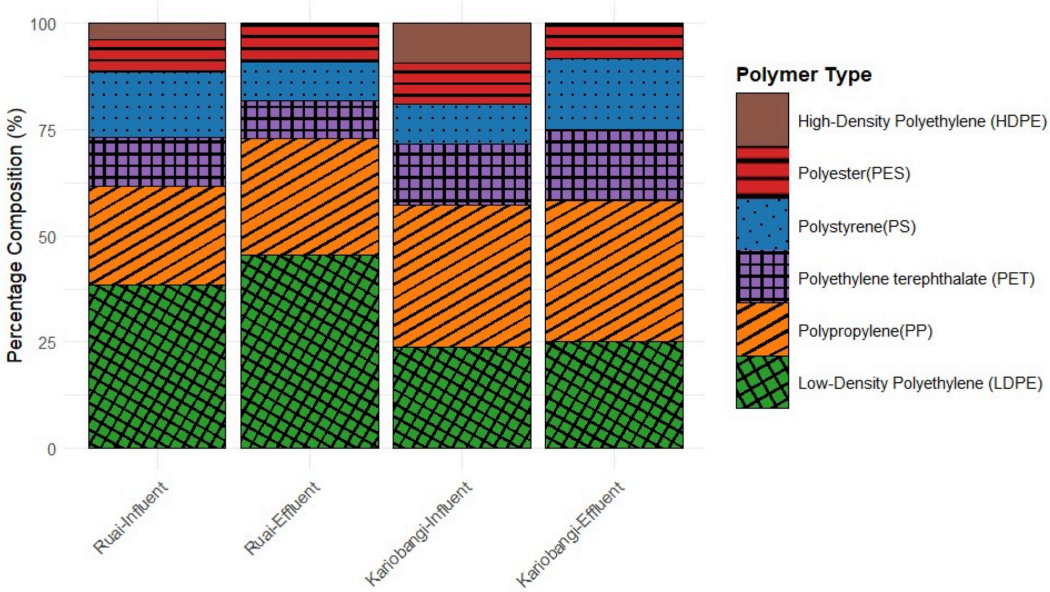

Figure 4.9: Percentage Composition of Different Polymers in the Influent and Effluent of the Ruai and Kariobangi Wastewater Treatment Plants

Polyethylene (PE) and polypropylene (PP) are among the most affordable and widely used types of plastic polymers, commonly found in many daily applications (Akarsu et al., 2019). PE is often used in personal care products, such as body and facial scrubs, soaps, toothpaste, and shampoos (Cristaldi et al., 2020; Liu et al., 2019; Mintenig et al., 2017). Additionally, it is utilised in food packaging, films, plastic bags, and various household items (Habib et al., 2020; Mintenig et al., 2017). PE can also originate from fibrous materials, such as clothing, fabrics, and carpets. On the other hand, PP primarily originates from the production and use of various plastic consumer products, mainly for packaging, food containers, and textiles (Habib et al., 2020). The high prevalence of both PP and PE in Nairobi can be attributed to the widespread use of consumer products made from these plastic materials.

The removal of HDPE from the treatment units in both treatment plants can be attributed to its elimination during the treatment processes. HDPE is a low-density polymer that tends to float, making it susceptible to removal during skimming operations. Additionally, the formation of biofilms as HDPE interacts with wastewater in the treatment units may increase its density, leading to sedimentation; as a result, it can become retained in the sludge (Ma et al., 2024).

A high presence of PE and PP has also been reported in influent and treated waters from various wastewater treatment plants. For instance, Akarsu et al. (2019) found PE (51%) and PP (35%) to be the predominant microplastics in wastewater treatment plants in Mersin Bay, Turkey. Similarly, Park et al. (2020) reported that PP (39.6%) and PE (25.6%) were the most abundant microplastics in wastewater treatment plants in South Korea. Blair et al. (2019) also reported PP (23%) and PE (4%) as the dominant polymers in a treatment plant in Scotland, UK. Additionally, Bayo et al. (2020) identified LDPE as the most prevalent type of particle, consisting of 70.61% of the total microplastics, in Murcia, Southeastern Spain. These results demonstrate the significant abundance of PE and PP, as well as their widespread use in consumer products.

4.3.5 Microplastic Removal Efficiency and Estimated Discharge

The Ruai wastewater treatment plant achieved a microplastic removal rate of 89 ± 1.91%, while the Kariobangi wastewater treatment plant achieved a removal rate of 78 ± 3.16% as shown in Table 4.4. An ANOVA test indicated a statistically significant difference in removal efficiency between the two plants (ANOVA, P= 0.006). The Ruai wastewater treatment plant was found to discharge an average of 123.1 ± 20 million microplastics per day into the Nairobi River. Kariobangi wastewater treatment plant, on the other hand, discharged approximately 10.3 ± 0.91 million microplastics per day during the studied period, as shown in Table 4.4.

Table 4.4: Average Water Volume Discharged, Percentage Removal Efficiency, and Average Number of Microplastics Released per Day in the Ruai and Kariobangi Wastewater Treatment Plants in Nairobi, Kenya

WWTP	Average volume discharged (m^3/day)	Removal Efficacy (%)	MP Released (particles/day)
Kariobangi	10,000	78 ± 3.16%	10.3 ± 0.91 million
Ruai	114,000	89 ± 1.91%	123.1 ± 20 million

The variation in removal efficiency between the two wastewater treatment plants may be influenced by the different treatment technologies and operational conditions (Conley et al., 2019; Cristaldi et al., 2020; Wang et al., 2020). Each plant employs distinct treatment techniques and has varying operational conditions for the treatment units, which likely contribute to the observed differences in removal efficiency. The Ruai wastewater treatment plant demonstrates higher removal efficiency due to its lagoon system, which provides a longer retention time. This extended retention time promotes biofilm formation and allows for the settling of heavier microplastics (Carr et al., 2016; Fan et al., 2023). In contrast, the relatively low removal efficiency at the Kariobangi wastewater treatment plant can be attributed to the declining performance of several treatment units. Many of these units are dysfunctional, and the plant is also facing issues such as overcapacity, ageing infrastructure, and limited pre-treatment capabilities, all of which likely hinder its ability to remove microplastics (Kilingo et al., 2021).

Despite the moderate removal rates and low concentrations of microplastics observed in the effluent, the total number of particles discharged is still significant. These facilities process millions of litres of wastewater daily, making the contribution considerable. The high levels of microplastics released highlight the urgent need for targeted treatment technologies to prevent microplastics from entering aquatic environments. Upgrading existing infrastructure with advanced treatment technologies, along with regular maintenance of treatment units, can improve efficiency. Additionally, regulatory standards should be strengthened, such as establishing a permissible limit for microplastics in effluents. The considerable volume of microplastics flowing into rivers demonstrates that wastewater treatment plants are a source of these pollutants into the environment (Gies et al., 2018; Magni et al., 2019; Murphy et al., 2016).

In comparison to microplastic concentrations reported in other studies, the removal efficiency observed in this study is considerably lower. However, the daily discharge levels are consistent with findings from some previous studies. For instance, Murphy et al. (2016) reported a microplastic removal efficiency of 98.41% at a wastewater treatment plant in Scotland, UK. They estimated that this corresponds to a daily discharge of 65 million microplastic particles. This higher removal efficiency was attributed to the use of flocculation agents during the treatment process, which facilitated the formation of flocs. Similarly, Blair et al. (2019) reported a microplastic removal efficiency of 96%, estimating that approximately 12 million microplastic particles are discharged daily from a wastewater treatment plant in Scotland, UK. The higher removal efficiency may be attributed to the implementation of the tertiary treatment process. Additionally, Akarsu et al. (2019) reported an average removal efficiency of 57% across three wastewater treatment plants in Mersin Bay, Turkey, with daily discharges reaching up to 180 million microplastic particles. Notably, one of the plants employing a tertiary treatment process still exhibited lower microplastic concentrations in the effluent compared to those using only secondary treatment processes.

These findings indicate that microplastic discharges are highly variable and suggest that the treatment technologies employed at each facility play a crucial role in their retention (Alvim et al., 2020). In contrast to this study, which relies on secondary treatment processes, these results underscore the need to upgrade existing treatment systems and integrate advanced tertiary filtration systems. Overall, these results, along with findings from other studies, suggest that wastewater treatment plants represent a significant pathway for microplastic pollution into the aquatic environment, due to the large volumes of effluent discharged daily. This poses a critical threat to the aquatic ecosystem and human beings.

CHAPTER FIVE

CONCLUSIONS AND RECOMMENDATIONS

5.1 Conclusions

This study investigated the concentration, composition, and distribution of microplastics in the Nairobi, Mathare, and Ngong Rivers, and evaluated the influence of human activities and the effectiveness of selected wastewater treatment plants in removing microplastics. The conclusions are summarised according to the study's specific objectives.

i. **Quantification of microplastic concentration and composition**
   - Microplastic abundance in the surface water of the rivers ranged from 0.22 ± 0.026 to 0.917 ± 0.061 particles/litre, with the highest concentrations recorded near industrial, commercial, and densely populated informal settlements.
   - The most common composition of microplastics detected were fibres (51.15%), white particles (40.08%) and polyethylene (PE, 43.33%).
   - The dominant types of microplastics were secondary microplastics, particularly fibres and fragments, suggesting that most pollution originates from the fragmentation of larger plastic items from nearby terrestrial sources.

ii. **Relationship between anthropogenic activities and microplastic abundance**
   - Land use analysis indicated a positive correlation observed between microplastic abundance and built-up areas, while forested areas and wetlands showed a negative correlation with microplastic levels.
   - These findings highlight the significant role of human activities, including residential, industrial, and commercial practices, in contributing to microplastic pollution in the rivers.

iii. **Assessment of WWTP efficacy in microplastic removal**

- The study found that microplastic concentrations in the influent of Ruai and Kariobangi wastewater treatment plants averaged 9.99 ± 0.50 particles/litre and 4.7 ± 0.28 particles/litre, respectively, which decreased to 1.08 ± 0.18 particles/litre and 1.03 ± 0.08 particles/litre in the effluent.
- This corresponds to average removal efficiencies of 78% and 89%, respectively. Despite these reductions, both plants still release a substantial number of microplastics into the environment (approximately 123.1 million and 10.3 million microplastics per day), indicating the need for improved treatment technologies.
- The most common types of microplastics found in both Ruai and Kariobangi WWTPs, in both influent and effluent, were fibres, transparent particles, and Low-density Polyethylene (LDPE) in Ruai, and polypropylene (PP) in Kariobangi WWTP.

5.2 Recommendations

5.2.1 Policy and Management Recommendations

i. Reduce plastic usage, improve recycling practices, and promote the reuse of plastic products to minimise the generation of secondary microplastics.
ii. Implement and enforce regulations on plastic waste disposal and management, particularly in densely populated and industrial areas, to reduce microplastic inputs into rivers.
iii. Upgrade wastewater treatment technologies to enhance microplastic removal efficiency and implement stringent monitoring to prevent the discharge of microplastics into aquatic environments.

5.2.2 Recommendations for Further Research

i. Conduct long-term studies across all seasons to investigate temporal variations in microplastic concentration and composition.

ii. Analyse microplastic distribution at different stages of wastewater treatment and in sludge to better understand removal mechanisms and identify opportunities for improved treatment.

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APPENDICES

Appendix I: NACOSTI Research Permit

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Appendix II: Approval for Collection of Wastewater Data

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11th March, 2025

Jelimo Phanice
Jomo Kenyatta University of Agriculture and Technology,
Po Box 6200-00200
Tel: (067)5870001
Email: sobee@jkuat.ac.ke

Dear Phanice,

RE: RESEARCH – THE EVALUATION OF ABUNDANCE AND SPATIAL DISTRIBUTION OF MICRO PLASTIC IN URBAN RIVERS

Reference is made to your letter dated 24th February 2025 on the above-mentioned subject.

Approval is hereby granted to you to collect data from **17th March, 2025** to **30th April, 2025** for your research project entitled **"The Evaluation of Abundance and Spatial Distribution of Micro Plastic In Urban Rivers."**

The Treatment Works Coordinator whose office is located at Ruiru Dam will assist you with the relevant Data/information in relation to your project.

All findings/information on Company matters should be accorded utmost confidentiality.

Please note upon completion, you will be expected to submit a copy of the findings to the office of the undersigned.

By a copy of this letter the following Officers are hereby informed accordingly.

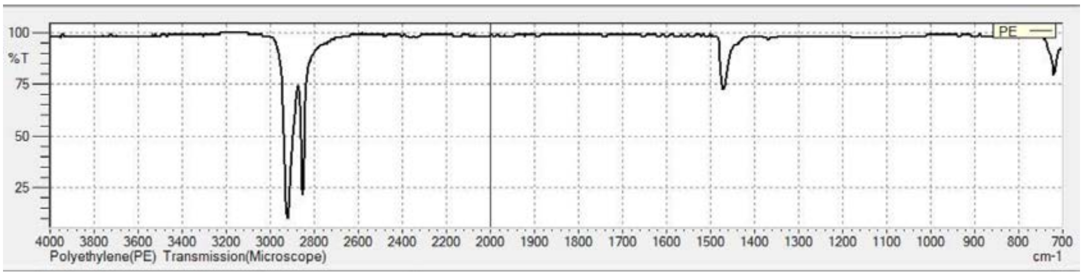
I. Treatment Works Coordinator – Ruai
II. Treatment Works Coordinator - Kariobangi
III. Research and Development Manager- National Water Offices

N.B: Kindly ensure you observe the Government directives on Covid -19 pandemic.

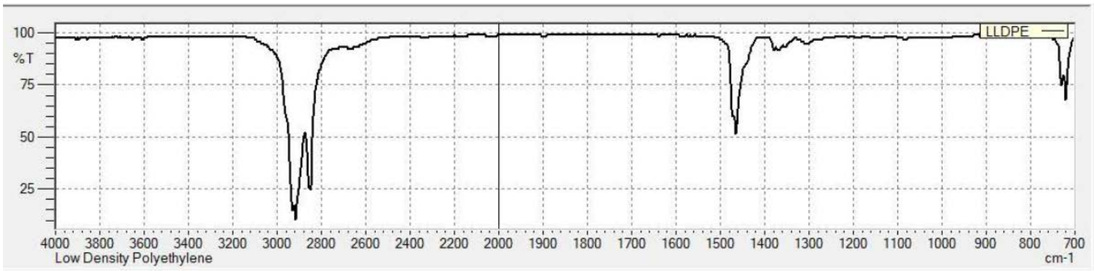
Yours Sincerely,

Martin W K Nang'Ole
For: Managing Director

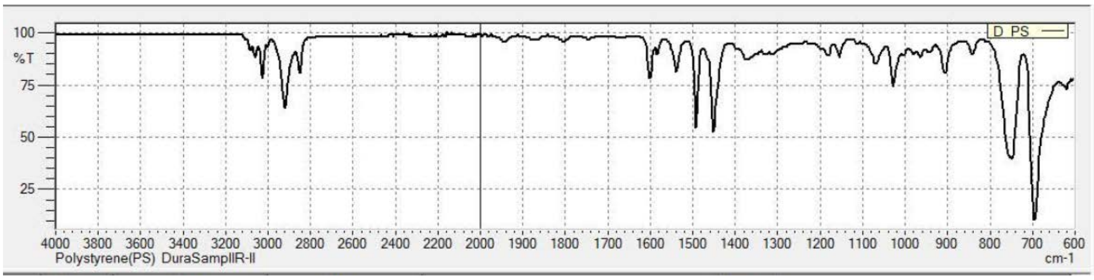
Appendix III: FTIR Spectra of Measured Microplastic Polymer in FTIR Spectroscopy



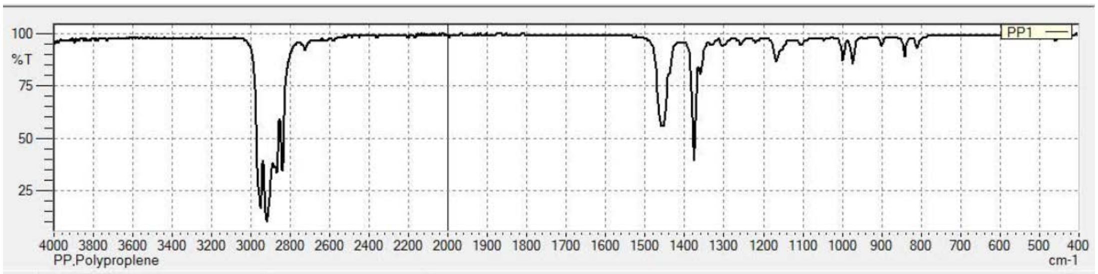
Spectral match; Polyethylene (PE)



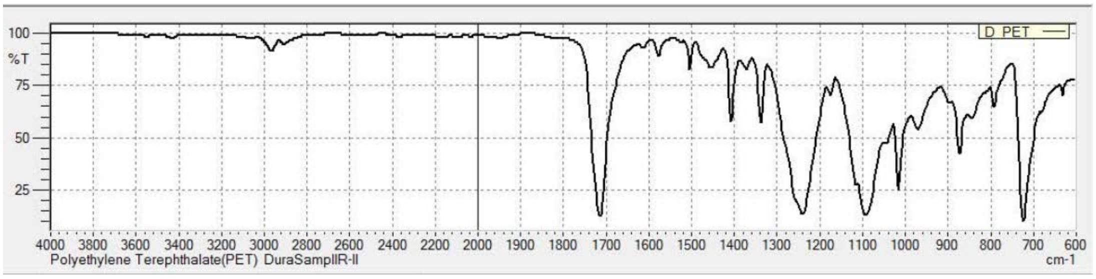
Spectral match; Low Density Polyethylene (LDPE)



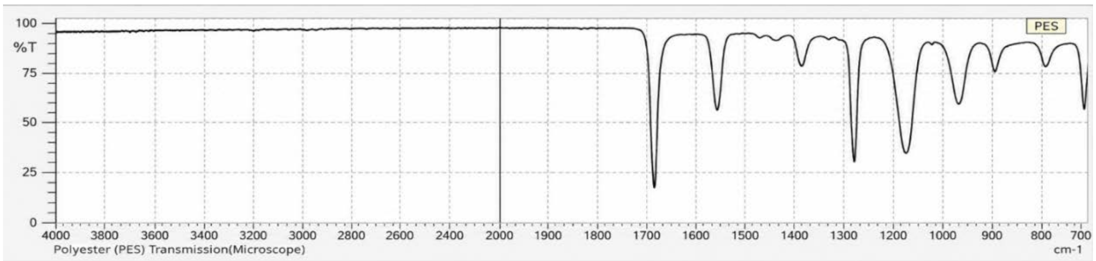
Spectral match; Polystyrene (PS)



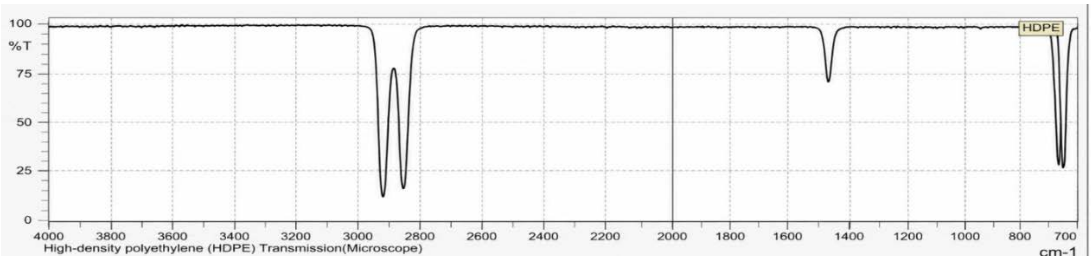
Spectral match: Polypropylene (PP)



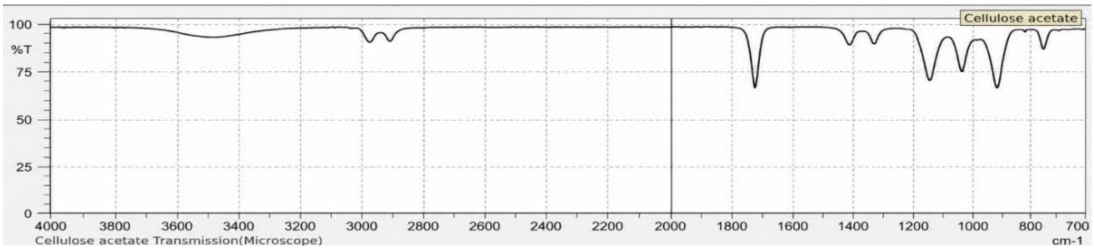
Spectral match: Polyethylene Terephthalate (PET)



Spectra match: Polyester (PES)



Spectra match: High-density polyethylene (HDPE)



Spectra match: Cellulose Acetate

Appendix IV: Photos from the Fieldwork

Appendix v: Photos from Laboratory Activities