

**IMPACT OF CATCHMENT LAND USE ON THE WATER QUALITY OF
WETLANDS IN MURANG'A COUNTY, KENYA**

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**A Thesis Submitted in Partial Fulfilment of the Requirements for the Degree of
Doctor of Philosophy in Chemistry of Murang'a University of Technology**

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DECLARATION

Student Declaration

I hereby declare that this thesis is my original work and to the best of my knowledge has not been presented for a degree awarded in this or any other university. I understand what plagiarism is and I am aware of the University policy in this regard. I understand that any false claim in respect of this work shall result in disciplinary action in accordance with the University Anti-Plagiarism Policy.

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DEDICATION

I dedicate this thesis to my beloved wife, Mrs. Viola Cherotich Yegon, and my children, Abigael, Yvonne, Lucky and Amani. It is through your support, devotion and noble succor that this academic journey has been successful.

I will forever remain grateful.

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ABSTRACT

Wetland ecosystems in Murang'a County are diminishing due to increased catchment land use practices. Agricultural practices carried out along wetland ecosystems involve excessive use of agrochemicals during crop production, contributing to wetland pollution via surface run-offs. This study was conducted to assess the impact of catchment land use on the water quality of Maragua and Mathioya riverine wetlands in Murang'a County, Kenya. The insights derived from this research are anticipated to contribute to policy formulation and sustainable management of wetlands. Both descriptive survey and experimental research designs were incorporated in this study. Questionnaires, containing items testing knowledge and attitude on the impacts of land use on the wetlands' water quality, were used to collect data from thirty sample of farmers. Several significant inter-question correlations were observed within the knowledge questions. Particularly, there is a negative correlation between Q5 and Q7 (-0.382 , $p = 0.037$), suggesting that farmers who know that wetlands drain into the River Mathioya are less likely to believe that the amounts of agrochemicals used can harm wetlands. Several significant inter-correlations among the attitude questions further illustrate the complexity of farmers' attitudes. Water and sediment samples were collected concurrently from upstream, midstream and downstream levels of the Maragua and Mathioya riverine ecosystems during both the dry and wet seasons. Elemental analysis was performed using Microwave Plasma-Atomic Emission Spectroscopy (MP-AES) instrument. Mean concentrations of lead during dry seasons were 0.147 ± 0.002 mg/L and 0.378 ± 0.004 mg/L for the wet season. Zinc were 0.01 ± 0.003 mg/L during dry season and 0.015 ± 0.004 mg/L during the wet season. Lead (Pb) concentrations surpassed the recommended WHO maximum residue level of 0.01 mg/L for drinking water. The mean of phosphate concentration was 0.0259 ± 0.0051 mg/L with a standard deviation of 0.0124 mg/L. Nitrate mean concentration during dry season was 9223.37 ± 2672.33 μ g/L with a standard deviation of 6545.84 μ g/L, and during wet season, it remained the same at 9223.37 ± 2008.17 μ g/L with a standard deviation of 4919.00 μ g/L. Pesticide residues were analyzed using HPLC-UV. Malathion residue mean concentration was 0.00575524 mg/L; 0.000580078 mg/L for thiamethoxam; 0.001241848 mg/L for diazinon while carbendazim was <LOQ across all the samples. All pesticide residues were below the recommended WHO maximum limits for drinking water. Thiametoxam showed a significant difference between sediment and water sample types with a p-value of 0.000 . Malathion concentration was asymmetrically distributed with a right tail; implying that a few samples contain significantly higher concentration than the average. Salinity, turbidity, total dissolved solids (TDS), electrical conductivity (EC) and p^H were analyzed across the sampling levels using hand-held portable water quality monitor meter. Salinity mean concentration across the three sampling levels was 116.28 ± 14.31 mg/L; 107.08 ± 13.32 mg/L for TDS; 0.16 ± 0.02 mS/cm for electrical conductivity (EC), turbidity: 160.38 ± 8.53 NTU and a p^H mean of 6.26 ± 0.09 . Variation in water parameters across the sampling levels showed that the wetlands have been polluted and the potential sources of pollution are agricultural run-offs and anthropogenic activities. The results highlight the need for integrated catchment management strategies that address land use practices to safeguard wetland health and ecosystem services in Murang'a County.

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ACRONYMS AND ABBREVIATIONS

APHA/AWWA/WEF	American Public Health Association/American Water Works Association/Water Environment Federation
BOD	Biochemical Oxygen Demand
DDT	Dichlorodiphenyltrichloroethane
DO	Dissolved Oxygen
DPSIR	Drivers, Pressure, State, Impact and Response
EC	Electrical Conductivity
EU	European Union
GAPs	Good Agricultural Practices
GC-MS	Gas Chromatography Mass Spectrometer
GOK	Government of Kenya
KALRO	Kenya Agricultural and Livestock Research Organization
LC-MS	Liquid Chromatography Mass Spectrometer
LSD	Least Significance Differences
MRLs	Maximum Residue Limits
NACOSTI	National Commission of Science Technology and Innovation
NPD	Nitrogen Phosphorus Detector
OCIs	Organochlorines
pH	Potential of Hydrogen
RSIS	Ramsar Sites Information Service
SAS	Scientific Analysis System
TDS	Total Dissolved Solids
UNEP	United Nations Environmental Protection
WHO	World Health Organization

CHAPTER ONE

INTRODUCTION

1.1 Background Information

Wetlands are defined as areas of land that are permanently or seasonally inundated with water, creating unique ecological conditions to which specific plant and animal species have adapted over time (GOK, 2015a). Such ecosystems are essential in maintaining environmental balance and are among the key productive ecosystems globally. Wetlands contribute to approximately one-third of all terrestrial primary production, highlighting their significant role in carbon cycling and ecological productivity. They serve as habitats for approximately seventy percent of the world's biodiversity, including numerous species of birds, fish, amphibians, and invertebrates, making them critical hotspots for conservation (Junk *et al.*, 2013).

Beyond biodiversity, wetlands are indispensable to human well-being and economic sustainability. They are key providers of a number of ecosystem services such as the supply of drinking water, food resources (e.g., fish and rice), flood regulation by acting as natural sponges, groundwater recharge, and supporting various cultural and harbor numerous plants with traditional medicinal uses (Bhowmik, 2022). In addition, wetlands contribute to climate regulation by storing carbon and moderating local temperatures, and they offer recreational and aesthetic benefits that enhance human livelihoods (Widney *et al.*, 2018). Their ability to filter pollutants and trap sediments significantly improves the quality of water downstream, which is crucial in maintenance of the integrity of freshwater systems as well as protecting human health (Widney *et al.*, 2018).

However, despite their ecological and economic importance, anthropogenic activities increasingly threaten wetlands. The rapid pace of urbanization, expansion of industrial infrastructure, deforestation, and unsustainable land use practices have contributed to a marked decline in the number and size of wetlands worldwide. Africa, in particular, has experienced significant wetland degradation due to these pressures. Notably, major wetland systems such as the Amazon and the Congo Basin, which account for over half of the world's wetland area, face mounting threats from land conversion and climate change (Mitsch & Gosselink, 2015). The ongoing loss of these critical

ecosystems underscores the urgent need for integrated management strategies and stronger local, regional, and global conservation policies.

In Kenya, wetlands predominantly exist in marshes, floodplains, and swamps and are vital in supporting ecological functions and local livelihoods (Oduor *et al.*, 2015). These wetlands cover approximately 3 - 4% of the national land area. However, during the rainy seasons—when heavy rainfall and surface runoff increase—the wetland coverage can expand to nearly 6% of the landmass as floodwaters spill into adjacent hinterlands, particularly in swampy and low-lying regions (Odour *et al.*, 2015; Macharia *et al.*, 2010).

Kenya's wetlands are categorized into six classes based on their hydrological, geomorphological, and ecological characteristics. These include human-made (artificial), lacustrine, riverine, marine, estuarine, and palustrine wetlands (Harmsen, 2018). Marine and estuarine wetlands are typically found along the coastal regions, while palustrine and lacustrine wetlands are prevalent in inland freshwater systems such as swamps, bogs, and lakes. Artificial wetlands, such as dams, ponds, fish farms, gravel pits, and rice paddies, are constructed primarily for economic or domestic use and now serve secondary ecological roles (Harmsen, 2018).

The geographical distribution of these wetlands reflects regional variations in rainfall, topography, and hydrology. Well-watered areas such as the highlands, coastal strip, and the Lake Victoria basin host the majority of natural wetlands. Riverine wetlands, in particular, are closely associated with major river systems. In Kenya, these include wetlands along the Tana, Athi, Yala, Ewaso Ng'iro, and Nyando rivers, which support a range of flora and fauna and are integral to water regulation and biodiversity conservation (Harmsen, 2018). Despite their ecological and economic importance, many wetland areas face increasing pressure from land-use change, pollution, and infrastructure development, necessitating targeted conservation and policy interventions.

Pollution is one of the most significant threats undermining wetland ecosystems' ecological integrity and functional usefulness. In Kenya and other parts of the world, the primary source of wetland pollution is closely linked to catchment land use practices, particularly those associated with agriculture (Abillah *et al.*, 2021).

Agricultural activities, often conducted along the fringes of small riverine and marshland wetlands, frequently involve extensive use of inorganic fertilizers and synthetic pesticides to enhance crop yields (Mwongeli, 2018). While these inputs may boost agricultural productivity, they pose substantial long-term risks to wetland health.

The runoff from these farmlands introduces excessive nutrients such as nitrogen and phosphorus into wetland waters, resulting in eutrophication, which depletes oxygen levels and significantly disrupts aquatic life. Moreover, the leaching of chemical residues, including pesticides and herbicides, into surface and groundwater systems contributes to chemical contamination, degrading water quality and posing health risks to aquatic and terrestrial organisms (Wagner *et al.*, 2008). In particular, the accumulation of agrochemical residues in wetland waters and soils has been associated with reduced biodiversity, altered food web dynamics, as well as loss of essential ecosystem services (Chang *et al.*, 2021; Momanyi *et al.*, 2019).

Beyond agricultural pollution, point source pollutants from pharmaceutical facilities, healthcare centers, domestic homesteads, and industrial operations further exacerbate wetland degradation. Wastewater and untreated effluents discharged from these sources often contain a complex mixture of harmful substances, including heavy metals, pathogens, and pharmaceutical compounds. Such pollutants disrupt the physicochemical properties of wetland water and impair vital environmental processes such as nutrient cycling and habitat provisioning (Maua *et al.*, 2022). Industrial effluents, in particular, frequently reach wetlands through stormwater and surface runoff, further compounding contamination levels and threatening human and environmental health (Mwangi, 2021). The cumulative effects of diffuse and point-source pollution call for urgent policy interventions, improved land-use planning, and the promotion of sustainable agricultural and industrial practices to safeguard Kenya's vulnerable wetland ecosystems from further deterioration.

Murang'a County, located in central Kenya, is characterized by small riverine and marshy wetland ecosystems that support local biodiversity, water regulation, and livelihoods. These wetlands are often situated along river basins such as the Maragua and Mathioya, where surrounding communities commonly carry out intensive agricultural practices (Mwangi, 2021). Conversion of wetland ecosystems to farmlands has contributed significantly to wetland degradation. Culturing crops along

riverbanks and in marshy zones typically involves the application of synthetic pesticides and inorganic fertilizers, which are often poorly regulated or managed. As a result, such practices have increased nutrient loading and chemical contamination within these wetland systems, undermining their ecological integrity and compromising critical ecosystem roles, for instance, purification of water, habitat support, and flood regulation (Muema *et al.*, 2019).

Although several research and conservation initiatives have been undertaken in central Kenya to address wetland degradation, these efforts have primarily focused on general wetland protection and community sensitization (Abillah *et al.*, 2021; Macharia *et al.*, 2010; Mwangi, 2021) . There remains a significant research gap in quantifying and monitoring the specific levels of elemental, nutrient and pesticide inflows into wetlands directly attributable to catchment land use practices. Understanding such dynamics is necessary for designing effective and efficient management interventions that align with sustainable land use while preserving wetland functionality. Without robust data on the extent and nature of pollution inflows, mitigation strategies may remain reactive and insufficiently targeted, limiting their long-term effectiveness in restoring and sustaining wetland ecosystems in regions such as Murang'a County.

1.2 Statement of the Problem

Small wetlands in Murang'a County and similar regions face considerable challenges stemming from intensive catchment land-use activities, particularly those related to agricultural production. These wetlands, especially riverine wetlands, are critical natural resources providing essential ecosystem services such as drinking water and irrigation for local communities. Agricultural activities conducted along riparian zones and water basins often involve extensively using synthetic pesticides and inorganic fertilizers to enhance crop yields. However, these agrochemicals frequently enter the wetland ecosystems through surface run-off, contributing substantially to pollution and the deterioration of water quality within these sensitive environments (Abillah *et al.*, 2021). The accumulation of agrochemical residues in water bodies compromises the chemical balance and adversely affects aquatic organisms, thereby threatening biodiversity and the overall health of the wetland ecosystems.

Moreover, chemical soil and water contamination represents one of the most pressing environmental concerns confronting wetlands currently. Build-up of residue inflows

from inorganic fertilizers and pesticides in wetland ecosystems, leads to eutrophication, toxic effects on aquatic fauna, and disruption of ecological functions. Compounding these environmental concerns is the widespread lack of public awareness regarding wetlands' multifaceted benefits, including flood regulation, water purification, and habitat provision. Inadequate enforcement of government policies and weak regulatory frameworks exacerbate the problem by allowing unchecked draining, filling, and encroachment of wetlands for agricultural expansion or other land uses (Macharia *et al.*, 2010). Without effective conservation measures and integrated land-use management, degradation of such important ecosystems is likely to continue.

1.3 Objectives

1.3.1 General objective

To determine the impact of catchment land use on the water quality of wetlands in Murang'a County, Kenya.

1.3.2 Specific Objectives

The specific objectives were to:

- (i) Establish the status of farmers' knowledge and attitudes on the impact of catchment land use on wetland water quality in Murang'a County, Kenya.
- (ii) Examine the effects of catchment land use on nitrates, phosphates, heavy metals, and pesticide inflows into wetlands in Murang'a County, Kenya.
- (iii) Determine the changes in wetland water quality parameters in relation to catchment land use practices.

1.4 Null Hypotheses

H₀₁: Farmers in Murang'a County do not have sufficient knowledge of the effects of catchment land use on wetland water quality.

H₀₂: There are no significant differences in the level of nutrients, heavy metals, and pesticide inflows across the sampling levels on small wetland ecosystems in Murang'a County.

H03: Land use practices do not affect water quality indicators (TDS, salinity, pH , turbidity, and EC).

1.5 Significance of the study

This study was undertaken to determine the effects of different land use systems on small wetland ecosystems in Murang'a County, Kenya. The insights derived from this research are anticipated to contribute to evidence-based policy formulation and strategic planning for the sustainable management of wetlands. Specifically, the findings and recommendations from this study are expected to inform targeted interventions aimed at restoring and maintaining the environmental integrity of wetland ecosystems, enhancing their long-term sustainability, and improving overall water quality. By elucidating the relationships between various land use practices and pollutant inflows, the study also contributes to the broader goals of promoting food security and safeguarding water-dependent species' health, including aquatic fauna and human populations that rely on wetlands for essential ecosystem services.

Moreover, findings from this research enriches scientific knowledge in place, regarding the ecological impacts of diverse land use systems, for instance, agricultural activities, human settlements, and industrial operations on pollution levels within small wetland ecosystems. These findings establish a critical foundation for designing and implementing effective mitigation and adaptation strategies to reduce pollution burdens in Murang'a's wetlands. Ultimately, this contributes to strengthening the ecological resilience of these vital ecosystems while simultaneously supporting their socio-economic functions and benefits for local communities.

1.6 Scope of the Study

This study was carried out in wetlands within Murang'a County, in central Kenya. Murang'a County exhibits complex climatic and geological characteristics, including variable rainfall patterns, sloped terrain, and heterogeneous soil compositions. These environmental factors influence the mobility, accumulation, and degradation of chemical contaminants such as pesticides and nutrients in surface water and sediment. For instance, certain geological formations may enhance pesticide retention in soils. In contrast, others may facilitate leaching into waterways, affecting contaminant levels' spatial and temporal variability across different wetland sites.

1.7 Limitations of the Study

A number of limitations were encountered in this study, one being difficulty in obtaining accurate and comprehensive data on agrochemical usage from local farmers. Many participants hesitated to disclose the specific types, quantities, or application frequencies of fertilizers and pesticides they used. This lack of transparency limited the study's ability to directly link land use practices with the concentration of contaminants detected in both the water and sediment samples. Consequently, while the study could identify patterns and concentrations of key pollutants, it could not fully ascertain their precise sources or quantify their contributions from various agricultural practices.

Addressing these limitations in future research will require the establishment of trust-based relationships with farming communities, coupled with participatory monitoring approaches that integrate scientific and local knowledge systems. Broader regional studies that include upstream and downstream catchment areas would also provide a more comprehensive understanding of contaminant dynamics across interconnected ecosystems.

1.8 Contributions of the Study

The information from this study seeks to fill some of these gaps by investigating how catchment land use patterns influence water quality in the small wetland ecosystems of Murang'a County. By examining the concentrations of nutrients, pesticides, and other pollutants, the study intends to determine the extent of contamination and link observed patterns to agricultural and other land-based activities. The insights gained will support the formulation of sustainable land and water management strategies that balance agrarian productivity with the conservation of wetland ecosystem services.

1.9 Organization of the Thesis

Chapter one (introduction) comprise of the background of the study,statement of the problem, objectives, significance of the study, scope and limitations of the study. This chapter also states contributions of the study. Chapter two (Literature review) presents a review of the literature on farmers' knowledge and attitudes towards impact of catchment land use on water quality in Murang'a wetlands; nutrient, elemental and pesticide inflow analysis in wetland waters, theoretical model adopted to guide in this research. Chapter three (methodology) outlines the study area, research design, data

collection and analysis procedures, and summary of the chapter. Chapter four (Results and discussion) entails the findings of the research, analysis and discussions, and summary of the section four. Chapter five entails conclusion and recommendations from the research findings.

CHAPTER TWO

LITERATURE REVIEW

2.1 Introduction

This chapter presents a comprehensive review of existing literature on environmental conservation with a particular focus on wetland ecosystems. It synthesizes scholarly work on catchment land use systems and their effects on various wetland water quality parameters. The importance of this review is to provide a foundation for the current study by highlighting relevant research findings, identifying emerging themes, and uncovering existing knowledge gaps that justify further investigation. The chapter begins with examining community knowledge, perceptions, and attitudes toward wetland ecosystems, recognizing the roles played by the local communities in conservation and sustainable utilization. It proceeds to explore wetland ecosystems' current status and degradation trends, followed by a detailed discussion on the nature, extent, and sources of pollution affecting these critical habitats. Subsequently, the chapter reviews various wetland conservation strategies and policy frameworks currently implemented in Kenya, evaluating their effectiveness and limitations. A conceptual framework is also presented to guide on the research methods applied and structure the analysis. Finally, the chapter concludes with a summary highlighting the existing gaps in literature, which this study aims to address through a focused investigation of small wetlands in Murang'a County.

2.2 Community Knowledge and Attitudes on Wetland Ecosystems

Community knowledge and attitudes toward wetland ecosystems are critical in ensuring proper management and sustainable conservation of these ecologically sensitive resources. The success of wetland protection and restoration efforts often hinges on local populations' awareness, involvement, and stewardship. Numerous studies have identified a direct correlation between community awareness on the residue inflow levels on wetlands, and the extent of wetland degradation or protection. Where local communities possess a limited understanding of wetlands' socio-economic and ecological importance, degradation is more pronounced, primarily as a result of unsustainable land-use decisions and behaviors driven by short-term human needs (Hertz-Picciotto *et al.*, 2018).

In Kenya and other developing countries, substantial portions of wetland ecosystems have been irreversibly converted into farmlands, grazing lands, and human settlement areas (Mwangi, 2021). This conversion is often driven by population pressure, land scarcity, and poverty, which force communities to prioritize immediate survival over environmental conservation. However, these actions overlook wetlands' long-term ecological services—crucial for environmental sustainability and human well-being (Robertson *et al.*, 2019). Wetlands serve essential roles in water purification by filtering pollutants, controlling floods by absorbing excess rainwater, sequestering carbon, and supporting diverse biological species. They also act as breeding grounds for fish and birds and a source of water for both domestic and agricultural use, especially in rural areas (Bhowmik, 2022).

Catchment land use practices, particularly those involving unregulated agricultural expansion and deforestation, have significantly compromised the stability of wetland ecosystems. For example, pollution from farmlands—primarily from the overuse of fertilizers and pesticides—leads to chemical runoff into wetland systems. This runoff has been shown to affect water quality indicators such as pH, salinity, dissolved oxygen (DO), biological oxygen demand (BOD), electrical conductivity (EC), and turbidity (Keché *et al.*, 2007). These changes in water quality can reduce aquatic biodiversity, promote the growth of invasive plant species, and ultimately compromise wetlands' ability to deliver ecosystem services.

In response to these challenges, global and local conservation efforts have been initiated to improve public understanding and promote the sustainable use of wetland resources. One prominent international initiative is the Ramsar Convention on Wetlands, established in 1971 to promote wise usage and preservation of wetland ecosystems. This international treaty has provided a legal framework which has ensured linkage between the national action and international cooperation aimed at ensuring conservation and sustainable usage of wetlands resources. Signatory countries, including Kenya, are encouraged to designate wetlands of global importance, implement strategic conservation plans, and to engage local communities in making decisions on processes related to wetland management (MEA, 2005).

Despite such global efforts, community knowledge and engagement in wetland conservation remains relatively low in many developing regions. Globally, especially

in sub-Saharan Africa, there still exists a significant gap in public awareness with regard to the ecological benefits of wetlands and the consequences of their degradation (Mitchell, S. A. (2013). This lack of understanding continues to hinder the effectiveness of wetland conservation policies and programs. Without a clear grasp of how overgrazing, unplanned urban expansion, or agrochemical usage affect wetlands, communities are unlikely to modify their behaviors or support conservation initiatives. This highlights the urgent need for context-specific educational interventions to sensitize communities on the important roles of wetlands, their threats, and the long-term benefits of conservation (Dube *et al.*, 2023).

In Kenya, various forms of commercial land use—such as intensive agriculture, hydroelectric power generation, industrialization, and rapid urbanization—have accelerated the degradation of wetlands. These land-use activities have led to severe consequences, including habitat loss, pollution effects from both the point and non-point sources, and also decline in critical ecosystem functions which include water purification, flood regulation, and biodiversity conservation (Twesigye *et al.*, 2011). In many cases, local communities who reside adjacent to wetlands are unaware that their seemingly benign activities contribute cumulatively to wetland degradation. Moreover, absence of proper land use planning and enforcement mechanisms allows encroachment and degradation to persist.

Murang'a County, located in Central Kenya, is a case in point. The county is endowed with numerous small wetlands that support local livelihoods and serve as essential catchment areas supplying water to neighboring counties, including Kiambu and Nairobi. These wetlands are critical for maintaining hydrological cycles, especially during drought or water scarcity. However, their ecological value and functional capacity have steadily declined due to increasing human encroachment, riparian degradation, and unsustainable land-use practices such as cultivation along riverbanks, sand harvesting, and deforestation (Muema *et al.*, 2019). Compounding the problem is the lack of adequate public education and engagement programs to build local capacity for sustainable wetland management.

Therefore, there is a pressing need for sustained community education and awareness campaigns that target both youth and adults. These programs should aim to instill a sense of environmental responsibility and stewardship among the public. Information

can be disseminated through schools, community forums, local radio programs, agricultural extension services, and non-governmental organizations (Rutter *et al.*, 2022). Additionally, participatory approaches that involve communities in wetland monitoring, conservation planning, and rehabilitation activities can foster a greater sense of ownership and accountability. Integrating traditional knowledge with scientific principles of conservation can also enhance the effectiveness of these initiatives, especially in rural and indigenous communities (Macharia *et al.*, 2010).

2.3 Degradation of Wetlands

Over the past five decades, various land use practices have greatly contributed to the widespread degradation of wetland ecosystems worldwide. Studies estimate that approximately 10% of global wetlands have been lost or severely degraded during this period, mainly as a result of anthropogenic activities such as agricultural expansion, urban development, industrialization, and infrastructure projects (Robertson *et al.*, 2019). This degradation has profound and multifaceted implications for the ecological integrity of wetland environments and human health, livelihoods, and overall well-being. Wetlands are essential in the provision of a wide range of ecosystem services—including water purification, flood regulation, carbon sequestration, and habitat provision—essential for sustaining biodiversity and human populations. Therefore, the loss or impairment of these services directly threatens the quality of life and resilience of communities that depend on them (Robertson *et al.*, 2019).

The degradation of wetlands manifests in several critical ways, notably through the decline in both the quality and quantity of water available and the loss of biodiversity. This has been documented in multiple studies, which link wetland deterioration with reductions in aquatic and terrestrial species richness and abundance, altering ecosystem functions and stability (Lishawa *et al.*, 2010; Albert *et al.*, 2020). Such ecological disruptions are associated with broad social, economic, and health challenges. For instance, diminished water quality can lead to increased incidence of waterborne diseases. At the same time, reduced biodiversity compromises the ecosystems' capacity to provide food, medicinal resources, and other goods crucial for human sustenance and economic activities (Lishawa *et al.*, 2010; Albert *et al.*, 2020).

The nexus between environmental sustainability and food security is powerful in sub-Saharan Africa. Wetlands are critical agricultural landscapes that support smallholder

farmers and rural communities, acting as buffers against climate variability by maintaining soil moisture and fertility. One of the significant consequences of unsustainable wetland development is the reduction in the physical extent of these ecosystems and the rapid degradation of soil quality within wetland areas. Soil fertility decline limits agricultural productivity, exacerbating food insecurity and poverty in vulnerable populations (Sakané *et al.*, 2011). This underscores the urgent need for integrated land use planning and wetland conservation efforts that balance development objectives with ecological sustainability to safeguard these vital resources for future generations.

Agricultural activities conducted in wetland areas often involve the extensive use of agrochemicals, including inorganic fertilizers, herbicides, and pesticides, to enhance crop yield and productivity. While these chemicals have improved agricultural output, their widespread and sometimes indiscriminate application has greatly contributed to the degradation and destruction of wetland ecosystems (Abillah *et al.*, 2021). Wetlands are highly sensitive environments where external inputs easily disrupt the water, soil, and biota balance. Introducing agrochemicals into these systems can alter water chemistry, reduce biodiversity, and impair the natural functions of wetlands, such as nutrient cycling and water filtration.

Satellite imagery and spatial analyses presented in the Kenya Wetlands Atlas provide compelling evidence of the changes occurring in wetland landscapes, revealing the extent of human-induced impacts. These changes are primarily driven by factors such as encroachment for agricultural and settlement purposes, hydrological modifications through drainage and water diversion, urban expansion, coastal zone degradation, and the broader effects of climate change (Sakané *et al.*, 2011). One of the most visible consequences of these processes is the conversion of large wetland masses into farmlands. This conversion has led to environmental challenges, including increased soil erosion, sediment displacement, and the gradual loss of wetland areas due to degradation (Abillah *et al.*, 2021).

Moreover, agrochemicals' excessive and often poorly managed application has further exacerbated wetland degradation. These chemicals frequently enter wetland systems through surface runoff, especially during heavy rains, contaminating water bodies within the wetlands. The runoff introduces high loads of nutrients, such as nitrogen

and phosphorus, along with pesticide residues, which can lead to eutrophication, reduced oxygen levels, and toxic effects on aquatic organisms (Mwongeli, 2018). Such contamination disrupts aquatic food webs, reduces species diversity, and threatens the overall sustainability of wetland ecosystems. Consequently, the ecological integrity of wetlands is compromised, undermining their ability to provide critical ecosystem services that benefit both the environment and human communities reliant on these resources.

Anthropogenic activities have been a major driver of wetland degradation worldwide, resulting in severe environmental consequences such as alterations in local and regional climate patterns and the loss of biodiversity, including the extinction of key wetland-dependent species (Robertson *et al.*, 2019). Wetlands, critical for maintaining ecological balance, have been fragmented and converted into agricultural farmlands, leading to habitat loss and disruption of natural hydrological regimes. Additionally, directly discharging untreated domestic sewage and industrial effluents into wetland systems has introduced toxic substances and excessive nutrients, further accelerating the degradation process (Ravikumar *et al.*, 2022). These pollutants impair water quality, reduce oxygen availability, and compromise the health of aquatic organisms, thus diminishing the overall ecological integrity of wetlands.

In Kenya, recent research conducted by the Kenya Wildlife Service on several important wetland ecosystems, including Lakes Nakuru, Bogoria, Naivasha, and Baringo, has highlighted alarming trends of declining water levels in these lakes. This decline is primarily attributed to increased rates of catchment encroachment for agricultural, settlement, and other land use purposes, reducing the wetlands' size and functionality (Weru, 2016). Such encroachment not only limits the capacity of these wetlands to buffer floods and filter pollutants but also threatens the sustainability of water supplies for surrounding communities and wildlife. The shrinking wetland areas exacerbate vulnerability to droughts and disrupt the habitats of numerous flora and fauna species that depend on these ecosystems for survival, thus underscoring the urgent need for integrated wetland management and conservation strategies.

2.4 Sources of Wetland Pollution

Wetland pollution is generally put into two main categories: point source and non-point source pollution. Each pollution type poses unique challenges to the ecological

integrity and sustainable management of wetland ecosystems. Point source pollution refers to contaminants originating from a specific, identifiable location. These sources typically include industrial effluent discharge pipes, wastewater treatment facilities, sewerage outlets, and drainage channels directly draining into wetland areas. Because the source is known and traceable, point source pollution is usually easier to regulate and monitor. However, the impact of point source pollution on wetlands can be severe. Discharging untreated or partially treated industrial and domestic wastewater introduces a broad range of hazardous pollutants into the wetland environment. These may include toxic heavy metals such as lead, mercury, cadmium, and chromium, as well as high concentrations of nutrients, pathogenic microorganisms, and organic waste materials (Ravikumar *et al.*, 2022).

Heavy metals, in particular, are of serious concern because they do not break down easily and tend to accumulate in wetland sediments and biological tissues. This accumulation can lead to bioaccumulation and biomagnification, where toxin concentrations increase up the food chain. These processes can result in severe health impacts on aquatic life, including reproductive issues, organ damage, and even death. For example, mercury exposure can impair neurological development in fish and amphibians, while lead can cause abnormal growth in aquatic plants and macroinvertebrates (Mwangi, 2016). In addition, untreated domestic sewage discharges elevate biochemical oxygen demand (BOD), causing decreased amounts of dissolved oxygen (DO) in the water. Low oxygen levels can suffocate aquatic organisms and disrupt the natural ecological balance. Moreover, pathogenic organisms introduced through sewage pose a health hazard to humans, particularly in communities depending on wetland waters for drinking, cooking, and bathing (Lu *et al.*, 2016).

On the other hand, non-point source pollution is more diffuse and originates from multiple, widespread sources across the landscape, making it significantly harder to monitor and manage. Agricultural runoff is one of wetlands' most prevalent sources of non-point pollution. This occurs when rainfall or irrigation water flows over agrarian lands, picking up pollutants such as inorganic fertilizers, pesticides, herbicides, animal waste, and sediments and transporting them into adjacent wetlands (Ravikumar *et al.*, 2022). In the East African region, including Kenya, agriculture has

been recognized as a major contributor to wetland degradation through non-point source pollution (Abillah *et al.*, 2021).

When agrochemicals like nitrogen- and phosphorus-based fertilizers enter wetland systems, they can result in nutrient build-up, a condition known as eutrophication. Eutrophication often contributes to rapid growth of algae and invasive aquatic plants, which can form a mass of mats on the water surface. Such mats interferes with sunlight penetration to submerged vegetation, reducing oxygen production and altering plant communities. Over time, this shift in plant species composition reduces the ecosystem's biodiversity. It undermines the services wetlands provide, for instance, water purification, flood regulation, carbon sequestration, and habitat for fish and wildlife (Robertson *et al.*, 2019).

In addition to surface runoff, agrochemicals can enter wetlands through groundwater infiltration. When chemicals applied to farmlands seep through the soil, they can contaminate the groundwater that feeds into wetlands. This indirect pathway is particularly problematic because groundwater pollution is less visible and often goes undetected until significant ecological damage has occurred. As groundwater moves slowly, pollutants can be environmentally persistent, affecting wetland ecosystems even after agricultural practices have changed (Mironga, 2005).

The cumulative effects of both point and non-point pollution are far-reaching. They compromise the ecological structure, functional capacity, and resilience of wetlands (Kimwaga *et al.*, 2012). Moreover, people depending on wetlands for their livelihoods, water supply, as well as cultural practices are directly impacted. For example, pollution-related fish kills affect biodiversity, disrupting food security and income generation for fishing communities. Similarly, the contamination of wetland water sources can lead to increased healthcare costs and reduced agricultural productivity when clean water is unavailable for irrigation or livestock (Anthonj *et al.*, 2019).

Addressing the twin challenges of point and non-point source pollution in wetlands requires a multifaceted approach. On the one hand, strengthening regulations and enforcement mechanisms is crucial to ensure that industries and municipalities treat their wastewater before discharge (Amondi *et al.*, 2005). On the other hand,

community education and engagement are vital for managing non-point source pollution, especially in rural agricultural areas. Educating farmers on best management practices (BMPs), such as controlled use of fertilizers, integrated pest management, proper waste disposal, and maintaining vegetative buffer zones along waterways, can significantly reduce the volume and toxicity of agricultural runoff (Shah, 2021).

Moreover, policy interventions that promote sustainable land use planning and incentivize environmentally friendly agricultural techniques can help reduce pollution loads entering wetlands. Government agencies and environmental organizations must also prioritize wetland monitoring programs to track pollution trends and inform adaptive management strategies. Technologies such as remote sensing, water quality testing kits, and GIS-based mapping tools can support more effective surveillance of pollution sources (Odha, 2024; Tonui, 2022); .

2.4.1 Pesticides

Synthetic pesticides are widely employed in agricultural practices to safeguard crops from a broad spectrum of pests and diseases, enhancing crop yield and food security. These pesticides are classified in consideration of their chemical formulation, nature, intended purpose, target organisms, mode of action, and level of hazardousness (Aktar *et al.*, 2009). Both inorganic pesticides, such as heavy metal compounds, and organic pesticides, including organophosphates and carbamates, are commonly applied in crop protection (Momanyi *et al.*, 2019). However, the continuous emergence of new pest and disease strains, alongside the development of resistance to existing pesticide formulations, has driven farmers to apply pesticides more frequently and in higher quantities. This intensive use contributes to the persistence of pesticide residues on crops and promotes contamination of adjacent riverine and wetland ecosystems through surface run-offs during rainfall events or irrigation (Ottiger *et al.*, 2019). Such pollution can harm aquatic ecosystems, leading to acute and chronic toxicity that causes poisoning, physiological stress, reproductive failure, and even mortality of marine organisms, including fish, amphibians, and invertebrates. Beyond ecological damage, pesticide contamination in water sources poses significant health risks to animals and human populations that rely on these waters for drinking, domestic use, and irrigation, potentially resulting in long-term effects such as endocrine disruption,

neurotoxicity, and carcinogenicity (Ottiger *et al.*, 2019). These findings shows the urgent need for integrated pest management practices and strict regulation of pesticide application to mitigate environmental contamination and protect both the ecosystem and public health.

Most synthetic pesticides are characterized by their high toxicity and significant environmental persistence, which often leads to their bioaccumulation in the tissues of animals across different trophic levels (Porta *et al.*, 2008). Bioaccumulation leads to the gradual build-up of toxins in the tissues of organisms, which can magnify through the food chain, affecting both target and non-target species. This accumulation poses severe adverse health effects, including neurotoxicity, endocrine disruption, reproductive impairments, and even carcinogenicity in exposed wildlife and humans (Porta *et al.*, 2008). The persistence of such pesticides in the environment leads to prolonged contamination of soils, surface waters, and groundwater systems, thereby impacting aquatic ecosystems and terrestrial wildlife habitats. These effects disrupt ecosystem functioning by altering species composition and reducing biodiversity, ultimately compromising essential ecosystem services to human well-being.

Due to the extensive ecological and human health risks associated with certain synthetic pesticides, international regulatory frameworks have been established to control or ban the use of the highly hazardous compounds. For example, organochlorine pesticides such as dichlorodiphenyltrichloroethane (DDT) have been banned or severely restricted in many developed countries due to their persistence, bioaccumulation potential, and toxicity (Kissane & Shephard, 2017; UNEP, 2004). The Stockholm Convention on Persistent Organic Pollutants (POPs), ratified globally, aims to eliminate or restrict the production and use of these harmful chemicals to safeguard environmental and human health. However, in some developing regions, such pesticides may still be used due to a lack of enforcement, limited access to alternatives, or economic constraints, which pose ongoing challenges for environmental management.

In response to these concerns, there has been a growing shift towards developing and adopting environmentally friendly pesticides. Biopesticides are formulated from natural products such as plant extracts, bacteria, fungi, or minerals, which tend to have low toxicity, rapid biodegradability, and minimal non-target effects (Porta *et al.*, 2008;

UNEP, 2003). Their use aligns with integrated pest management (IPM) approaches, emphasizing ecological sustainability and reduced chemical inputs. Environmentally friendly pesticides help to mitigate pollution risks in agricultural landscapes, reducing contamination of wetlands, rivers, and soils while maintaining crop productivity. Moreover, these biopesticides often target specific pests, thus preserving beneficial organisms that support natural pest control.

Despite their advantages, the large-scale adoption of environmentally friendly pesticides faces challenges, including higher costs, limited shelf life, variable efficacy under field conditions, and gaps in farmer awareness and technical knowledge (Porta *et al.*, 2008). Ongoing research is critical to improving formulation stability, broadening pest control spectra, and ensuring that these alternatives can replace hazardous synthetic pesticides effectively. Additionally, robust policy frameworks and extension services are required to support farmers transitioning to sustainable pesticide practices. Monitoring and evaluation programs are also essential to track both health and environmental impacts of pesticides, ensuring regulatory compliance and the protection of vulnerable ecosystems such as wetlands.

In conclusion, while restricting highly toxic synthetic pesticides represents significant progress in safeguarding environmental and public health, promoting and integrating environmentally friendly pesticides into agricultural systems are vital for achieving long-term sustainability. This dual approach helps to reduce ecological contamination and supports the resilience of ecosystems like wetlands that are particularly sensitive to chemical pollution. The successful management of pesticide use will require multidisciplinary efforts involving scientific research, policy development, farmer education, and community engagement.

2.4.2 Inorganic Fertilizers

Fertilizers are essential substances used to enrich soil with vital nutrients, enhancing soil fertility and improving crop yields. Due to the projected global population rise to approximately 9 billion by 2050, there is an increasing demand for food production, necessitating intensified use of fertilizers within the agricultural sector to meet these food security needs (Godfray *et al.*, 2010). Among the various types of fertilizers, synthetic inorganic fertilizers—such as urea, diammonium phosphate (DAP) are the most widely applied due to their immediate nutrient availability and effectiveness in

promoting plant growth (Ramteke & Shirgave, 2012). These fertilizers supply key macronutrients like nitrogen, phosphorus, and potassium, critical for plant development and high crop productivity.

Synthetic inorganic fertilizers, are also a potential source of heavy metals which are associated with significant environmental hazards. Nutrient overloading from fertilizers can lead to the eutrophication of nearby aquatic ecosystems, including wetlands, rivers, and lakes, thereby disrupting the delicate ecological balance (Hasler, 2017; Pretty & Hine, 2011). Excess nitrogen and phosphorus compounds, when transported via surface runoff from agricultural fields, increase nutrient inflows into wetlands, resulting in oxygen depletion, algae blooms, and loss of aquatic biodiversity (Momanyi *et al.*, 2019). These changes subsequently affect water quality, degrade habitats for marine species, and reduce the overall ecosystem services wetlands provide, such as water purification and flood regulation.

In addition to ecological impacts, the mismanagement and over-application of fertilizers in farming—particularly in developing countries—pose substantial risks. Many smallholder farmers do not consistently adhere to Good Agricultural Practices (GAPs), such as applying fertilizers at recommended rates, timings, and methods, which leads to inefficient use of inputs and elevated environmental pollution (Momanyi *et al.*, 2019). Limited access to extension services, insufficient farmer education, and economic constraints often contribute to these challenges. The indiscriminate use of fertilizers increases production costs and exacerbates soil degradation and pollution of surrounding wetlands and water bodies.

To address these concerns, advanced agricultural technologies that emphasize environmental sustainability are increasingly encouraged. These include precision agriculture techniques that optimize fertilizer application, organic fertilizers and biofertilizers that minimize chemical inputs, and integrated nutrient management practices that combine organic and inorganic sources (Hasler, 2017; Pretty & Hine, 2011). Such approaches reduce environmental nutrient losses while maintaining or enhancing crop productivity. Moreover, promoting GAPs through farmer training, policy support, and effective regulatory frameworks is critical to minimize nutrient runoff into sensitive ecosystems such as wetlands.

In conclusion, while fertilizers remain vital for meeting the food production demands of a growing global population, their use must be carefully managed to prevent adverse environmental impacts. Enhancing farmer awareness and adopting environmentally friendly practices and technologies is essential to safeguard wetland ecosystems from nutrient pollution and promote sustainable agricultural development.

2.4.3 Heavy Metals Pollution

Heavy metals are a group of metallic elements and metalloids known for their environmental persistence and potential toxicity to living organisms. Scientifically, heavy metals are defined as metals with an atomic number greater than 20 and a density exceeding 5 g/cm³, characteristics that contribute to their chemical stability and tendency to bioaccumulate in ecosystems (Ali & Khan, 2018). These elements include but are not limited to, lead (Pb), cadmium (Cd), mercury (Hg), arsenic (As), chromium (Cr), and nickel (Ni), which pose significant risks to both aquatic and terrestrial biota when present in elevated concentrations.

Heavy metals enter wetland ecosystems through both natural and anthropogenic pathways. Natural sources include geological processes which includes weathering of rocks and soil erosion, which release metals into surrounding environments (Maier *et al.*, 2014). However, anthropogenic activities have increasingly become the dominant source of heavy metal pollution in wetlands. Industrial discharges, including effluents from mining operations, manufacturing plants, and waste treatment facilities, contribute substantially to the heavy metal load in aquatic environments. Additionally, agricultural practices contribute heavy metals by applying pesticides, fertilizers, and sewage sludge, which may have trace amounts of these metals (Maier *et al.*, 2014). Domestic wastewater and untreated sewage also release heavy metals directly into wetland systems, particularly in areas lacking adequate sanitation infrastructure.

The accumulation of heavy metals in wetland sediments and waters is a primary environmental concern because of their toxicity, persistence, and ability to bioaccumulate in aquatic organisms, thereby entering food chains and posing risks to human health (Ali & Khan, 2018). Wetlands act as sinks for heavy metals, trapping these contaminants in sediments where they can cause long-term ecological harm. Elevated heavy metal concentrations can disrupt microbial activity, reduce biodiversity, and impair ecosystem roles which include water purification and

provision of habitat. In aquatic organisms, heavy metals interfere with physiological processes, leading to reduced growth, reproductive failure, and mortality.

Given wetlands' ecological importance and vulnerability to pollution, understanding the sources, pathways, and impacts of heavy metals is crucial for effective management and remediation efforts. Monitoring heavy metal concentrations in wetland environments provides critical data to inform policies to reduce pollution inputs and protect biodiversity and human health.

Raw industrial sewage discharges are a major contributor to the increasing levels of heavy metal pollution in aquatic ecosystems globally (Abdeldayem, 2020). Industrial effluents often contain a complex mixture of heavy metals such as lead (Pb), cadmium (Cd), mercury (Hg), zinc (Zn), and copper (Cu), which enter water bodies untreated or partially treated, leading to the contamination of aquatic habitats. This pollution poses significant environmental and public health challenges, especially for species that depend on these ecosystems, including fish and humans (Mance, 2012). Heavy metals tend to bioaccumulate in aquatic organisms, particularly fish, concentrating in tissues such as muscles, liver, and gills. This bioaccumulation produces toxic effects, including physiological stress, impaired growth, and increased mortality rates (Authman *et al.*, 2015). Due to their ability to accumulate heavy metals, fish are reliable bio-indicators for assessing the extent and impact of heavy metal contamination in aquatic environments by exhibiting measurable physiological and biochemical changes (Authman, 2008; Rashel, 2001).

For humans, exposure to heavy metals through consuming contaminated fish and direct contact with polluted water has been linked to numerous health problems. Chronic exposure to these toxic metals is associated with immune system suppression, genotoxicity (damage to genetic material), neurotoxicity that affects the nervous system, and increased risk of chronic disease and mortality (Low & Higgs, 2014). Furthermore, heavy metals have been implicated in causing physical deformities, altered body morphometrics, and carcinogenic effects, resulting in long-term and sometimes irreversible health complications. Given these profound implications, monitoring and regulating industrial discharges is critical to safeguarding aquatic biodiversity and protecting human health from the adverse effects of heavy metal pollution.

Apart from using fish as bio-indicators to assess the effects of heavy metal pollution, several other parameters have been employed to monitor and evaluate contamination levels in wetlands and aquatic ecosystems. One such approach involves the assessment of microbial parameters in soils. Microbial activities such as nitrogen fixation, nutrient mineralization, and soil respiration are essential for maintaining soil health and nutrient cycling. Changes in microbial activity and population dynamics provide vital information on the extent and impact of heavy metal pollution in soils, as heavy metals can inhibit microbial processes and reduce microbial diversity, thereby impairing soil fertility and ecosystem functioning (Brookes, 1995).

In addition to biological indicators, geophysical methods such as the magnetic susceptibility technique have emerged as cost-effective and efficient tools for mapping the spatial distribution of heavy metal pollution. Magnetic susceptibility measures the degree to which soil and sediment particles can be magnetized and has been shown to correlate with the presence of industrial waste and atmospheric pollutants, which frequently contain magnetic particles associated with iron-bearing heavy metals (Lu *et al.*, 2007). This method allows for rapid, non-destructive identification of pollution hotspots and provides a valuable alternative or complement to traditional chemical analyses. Magnetic susceptibility mapping of soils and sediments has been successfully applied to estimate anthropogenic pollution levels and trace industrial contamination in diverse environments, thereby aiding environmental monitoring and management efforts (Hanesch & Scholger, 2002; Hanesch *et al.*, 2003).

2.5 Wetland Conservation in Kenya

Wetland conservation efforts in Kenya have been significantly advanced through coordinated initiatives led by the Kenyan Government, primarily facilitated by the Ministry of Environment in partnership with international organizations such as the United Nations Environment Programme (UNEP) and supported by a comprehensive policy framework (UNEP, 2018). This policy framework integrates constitutional provisions, international environmental conventions, and the roles of various government agencies, including the National Environment Management Authority (NEMA) and specialized ecological research institutes, all working collaboratively to safeguard and restore wetland ecosystems (Harmsen, 2018). These environmental policies aim to promote sustainable use, conservation, and rehabilitation of critical

environmental assets to ensure the long-term integrity and functionality of natural ecosystems.

A landmark commitment demonstrating Kenya's dedication to wetland conservation was its accession in the late 1990s to the Ramsar Convention on Wetlands of International Importance, a globally recognized treaty focused on the preservation and sustainable use of wetlands (Harmsen, 2018). Under the Ramsar Convention framework, several key wetland areas in Kenya have been designated as Ramsar Sites, underscoring their ecological significance and the need for protection. These include the Tana River Delta, Lake Nakuru, Lake Baringo, Lake Elementaita, Lake Naivasha, and Lake Bogoria, collectively covering approximately 265,449 hectares (Ramsar, 2018). These designations highlight the ecological and biodiversity value of these wetlands and reinforce Kenya's commitment to fulfilling international conservation obligations and promoting sustainable development through integrated wetland management strategies.

At the national level, wetland management in Kenya is governed by a robust legal and institutional framework designed to ensure the sustainable use and protection of these vital ecosystems. The Constitution of Kenya provides the foundational legal basis for environmental stewardship, emphasizing sustainable development and environmental conservation (GOK, 2010). Complementing the Constitution, specific legislation such as the Environmental Management and Coordination Act (EMCA) (GOK, 2015a) and the Wildlife Conservation and Management Act (GOK, 2015c) establish comprehensive guidelines and regulatory measures that directly address the conservation, utilization, and management of wetlands and their associated biodiversity.

These legislative instruments are further reinforced by sector-specific laws that regulate land use and spatial planning, notably the Land Act (GOK, 2012) and the Physical Planning Act (GOK, 2015b). These laws play a critical role in controlling activities within wetland catchments, thereby mitigating the adverse effects of unsustainable land-use patterns such as encroachment, agricultural expansion, and urban development. To operationalize these policies, the National Environment Management Authority (NEMA) has developed a suite of regulations and guidelines to safeguard wetland ecosystems. NEMA's initiatives focus on maintaining the

ecological functions of wetlands, which include supporting biodiversity conservation, sustaining hydrological cycles, and providing essential ecosystem services such as food provision, water filtration, and habitat for numerous species (NEMA, 2017). These frameworks collectively reflect Kenya's commitment to integrated environmental management and preserving wetland resources for present and future generations.

2.6 Parametric Analysis of Heavy metals

Parametric analysis of heavy metals in water samples from Kenya's wetlands has been the focus of several environmental studies aimed at evaluating water quality and associated health risks. These analyses typically involve quantitative determination of metal concentrations—such as lead (Pb), cadmium (Cd), copper (Cu), zinc (Zn), and chromium (Cr)—using techniques like Atomic Absorption Spectrophotometry (AAS) or Inductively Coupled Plasma Optical Emission Spectrometry (ICP-OES) and Microwave Plasma Atomic Emission Spectrometry (MP-AES).

A notable study conducted on the Sosiani River, which interacts with wetland ecosystems in Eldoret, found lead concentrations averaging 0.106 mg/L—exceeding both the WHO guideline of 0.01 mg/L and the US EPA maximum contaminant level (MCL) of 0.015 mg/L, indicating significant contamination (Jepkoech *et al.*, 2013). Copper and zinc levels in the same study were 0.29 mg/L and 0.41 mg/L, respectively, falling below WHO and EPA health-based limits (2.0 mg/L for Cu, no health-based limit for Zn) but with zinc surpassing EPA's secondary aesthetic guideline of 5 mg/L. In contrast, water samples from the River Isiukhu, a tributary within the Lake Victoria Basin wetlands, showed heavy metal concentrations were within WHO and EPA permissible limits, with lead, cadmium, and chromium often below detection limits (Raburu *et al.*, 2019).

Similarly, analyses from the Lake Naivasha wetland system revealed low concentrations of heavy metals in water, all within safe drinking water thresholds, though higher accumulation was observed in sediments and biota (Mutia *et al.*, 2012). These findings underscore spatial variability in contamination levels, with certain wetland-connected areas exhibiting concerning pollution, especially with respect to lead, while others remain within global safety standards.

2.7 Theoretical Framework

This study posits that diverse land use practices have significantly contributed to the pollution of wetland ecosystems in Murang'a County, Kenya, with agricultural activities being a primary driver. The increasing population growth in the region has intensified pressure on wetland resources, leading to unsustainable exploitation and pollution. Furthermore, limited awareness and inadequate implementation of Good Agricultural Practices (GAPs) among local farmers exacerbate the pollution levels, as improper use of agrochemicals and poor land management result in runoff carrying nutrients and chemical residues into wetlands (Momanyi *et al.*, 2019; Mwangi, 2021). Such pollution leads to the degradation of wetland ecosystems, as evidenced by diminished aquatic biodiversity, compromised water quality, and elevated concentrations of nutrients and toxic substances. These environmental changes threaten the ecological integrity of wetlands, pose significant health risks to surrounding communities, and contribute to broader ecological challenges such as climate change (Robertson *et al.*, 2019; Albert *et al.*, 2020). Given the wetlands' critical role in supporting livelihoods and maintaining ecosystem services, the escalating pollution in Murang'a County's wetlands calls for urgent intervention. Immediate mitigation measures, including community education, enforcement of environmental regulations, and promotion of sustainable land use practices, are necessary to preserve these vital ecosystems and safeguard their long-term functionality.

This study proposes several mitigation strategies to reduce pollution and enhance wetland ecosystems' sustainability in Murang'a County. Key measures include promoting alternative agrochemicals with lower toxicity and environmental persistence, thereby minimizing harmful residues entering wetland waters (Porta *et al.*, 2008; UNEP, 2003). Additionally, effective surface water management practices, such as controlling runoff through buffer zones and contour farming, are recommended to reduce the influx of nutrients and pollutants into wetlands (Momanyi *et al.*, 2019). Farmer education on Good Agricultural Practices (GAPs) remains critical to correctly applying agrochemicals and sustainable farming methods, which can mitigate detrimental impacts on wetland ecosystems (Mwangi, 2021). Furthermore, reforming land use policies to integrate environmental protection and conservation principles is essential for safeguarding wetland integrity while

accommodating agricultural development (NEMA, 2017). To systematically assess the interactions between land use and wetland water quality, this study adopted the Drivers, Pressure, State, Impact, and Response (DPSIR) theoretical framework conceptualized by Kristensen (2004). Figure 2.1 illustrates the DPSIR theoretical framework employed to evaluate how catchment land use dynamics influence the water quality of small wetland ecosystems in Murang'a County, Kenya. This framework facilitates a holistic understanding of the cause-effect relationships and informs targeted interventions for wetland conservation.

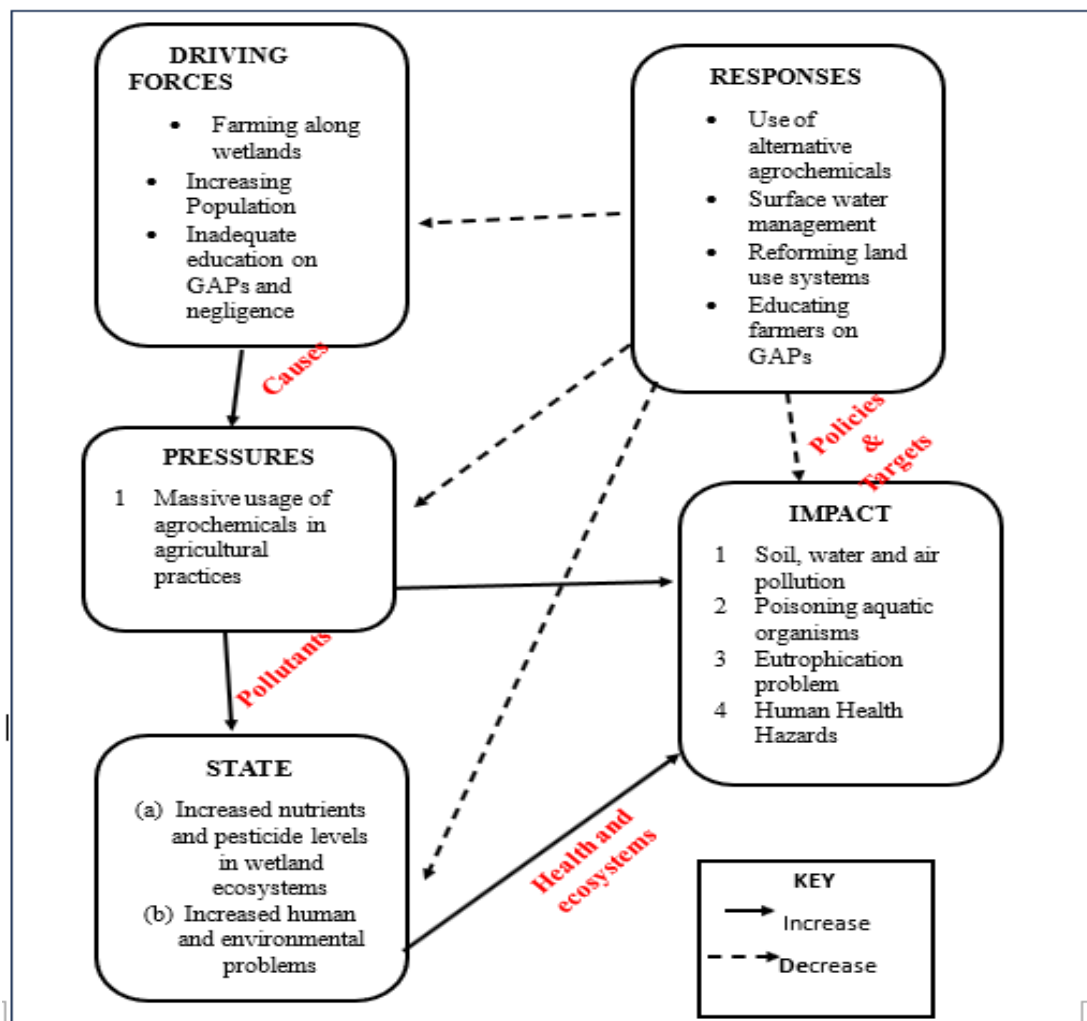


Figure 2.1: Theoretical Framework

2.8 Summary

This chapter has critically reviewed existing literature on wetland ecosystems, focusing on community knowledge and attitudes, the current condition of small wetlands about agricultural and anthropogenic activities, sources and types of wetland pollution, and conservation strategies implemented in Kenya. Additionally, it outlined

the theoretical framework adopted in this study: the Drivers, Pressure, State, Impact, and Response (DPSIR) model developed by Kristensen (2004). The reviewed studies consistently reveal that agricultural practices along wetland ecosystems predominantly involve agrochemicals, particularly inorganic fertilizers and synthetic pesticides, aimed at enhancing crop protection and yield. However, improper management and application of these agrochemicals result in significant wetland pollution, compromising water quality and disrupting the ecological balance of these sensitive ecosystems. This underscores the critical need for sustainable agricultural practices and effective pollution control measures to safeguard wetland health.

CHAPTER THREE

METHODOLOGY

3.1 Introduction

This chapter outlines all the materials, chemicals, and methods used in this study. It describes the location of the study area, selection criteria of the study sites, research and sampling designs, sample preservation and analytical techniques, analysis of pesticide residues in water and sediment samples, analysis of metal ions in water, data analysis and ethical statement. The chapter also outlines the analytical Quality Assurance for water and sediment samples applicable during pesticide residue analysis and provides a summary of the entire chapter.

3.2 Location of the Study Area

The Maragua and Mathioya River Basins, located in Murang'a County, Kenya, are notable for their ecological and socio-economic significance, particularly in supporting local agriculture and biodiversity. The Maragua River Basin is a small riverine wetland ecosystem situated in Murang'a South Sub-County, in the vicinity of Maragua Town, at geographical coordinates longitude 36.97°E, latitude 0.77°S, and an elevation of approximately 1600 meters above sea level (see figure 3.1). This basin comprises marshy wetlands and meandering tributaries that play a critical role in the hydrological and ecological dynamics of the region. It supports agricultural activities by providing a reliable water source and fertile soil for crop cultivation.

The Mathioya River Basin, on the other hand, is located in Mathioya Sub-County and shares borders with Nyeri County to the north, Kangema Sub-County to the northeast, Murang'a East Sub-County to the east, and Nyandarua County to the west. Like Maragua, the Mathioya basin features numerous small riverine wetlands essential for irrigation, soil fertility, and sustaining aquatic and terrestrial biodiversity. These basins lie within the larger Tana River catchment, a key watershed for central Kenya.

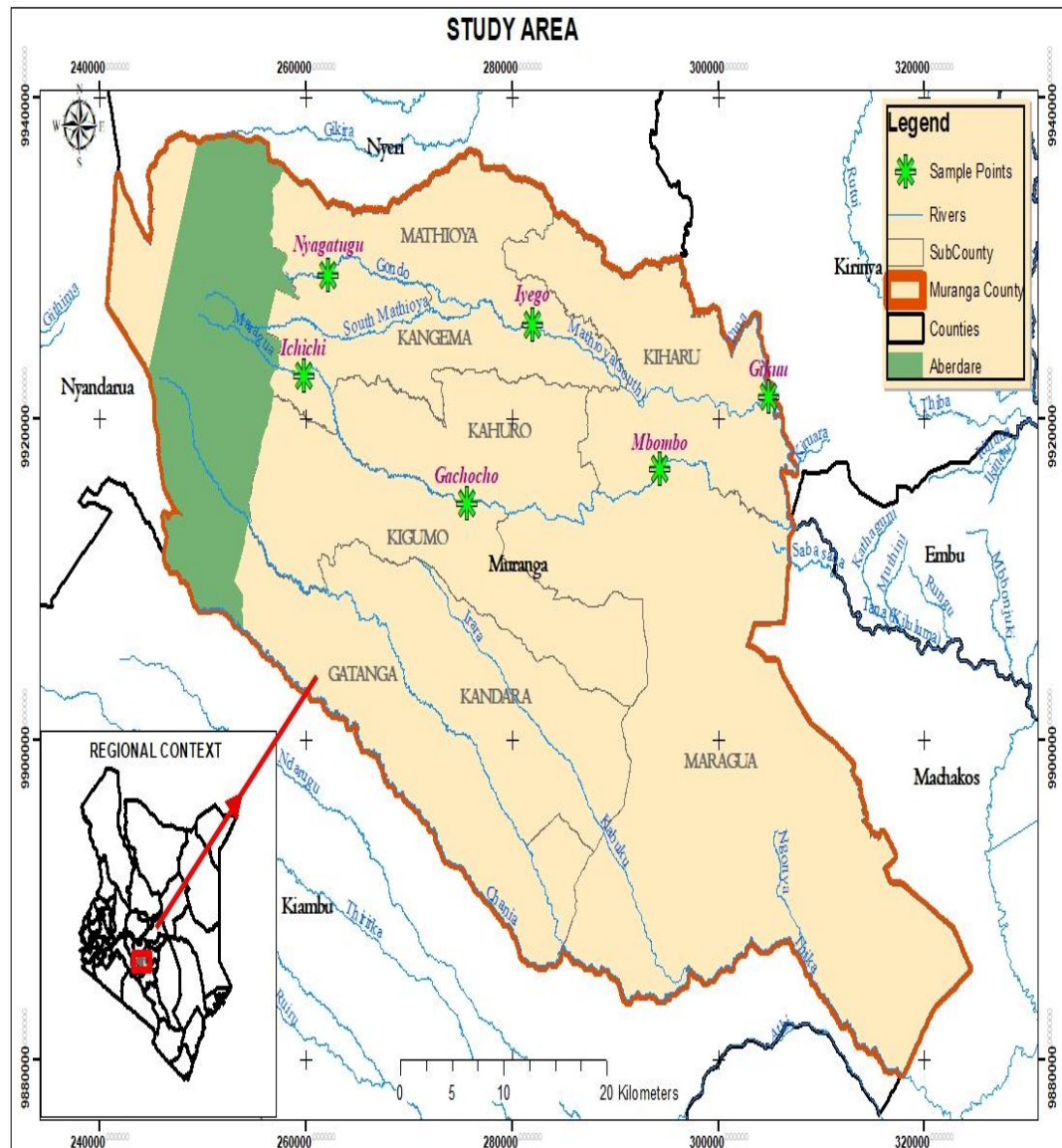
Murang'a County's primary economic activity is farming, supported mainly by deep red soils derived from volcanic materials resulting from extensive weathering of volcanic rocks, particularly in the upper regions near the Aberdare Mountains (Fairburn, 1966). The lower areas of the county are underlain by basement system rocks, contributing to a varied geological landscape. Murang'a County has a

population of approximately 1,056,640 people, with a substantial proportion residing in rural areas and depending on natural ecosystems for their livelihoods (KPHC, 2019).

The region experiences a bimodal rainfall pattern with approximate annual precipitation of 996 mm, with long rains between March and May and short rains between October and December (Ovuka, M., & Lindqvist, S. (2000). These seasonal rains are critical for replenishing wetland ecosystems and supporting rain-fed agriculture. The Maragua and Mathioya River Basins, with their marshy zones and consistent water flow, play an indispensable role in enhancing local food security, livelihoods, and ecosystem resilience.

Maragua and Mathioya River Basins, are situated within the Upper Tana River Basin, one of Kenya's most significant watersheds. The Tana River Basin extends for approximately 1,000 kilometers, originating from the Aberdare Ranges, and plays a central role in supplying water for domestic, agricultural, and industrial use across multiple counties, including Murang'a, Kirinyaga, Embu, and Nairobi. These river basins form part of the upstream catchment that contributes significantly to the overall flow and water quality of the Tana River, making their ecological integrity vital to downstream users.

The Maragua River Basin in Murang'a South Sub-County is a key sub-catchment area within this broader watershed. It borders four sub-locations: Ichagaki, Nginda, Gakoigo, and Maragua, and features a network of small wetlands and tributaries that feed into the main Maragua River. Over the years, human settlements and livelihood activities have increasingly encroached into riparian and water catchment areas. The surrounding communities have converted a significant portion of the wetland buffer zones into farmlands, primarily for subsistence and small-scale commercial agriculture (Muema *et al.*, 2018).



3.3 Research Design

This research employed a combination of descriptive survey and experimental research designs to comprehensively investigate the relationship between catchment land use and wetland water quality in the Maragua and Mathioya River Basins in Murang'a County, Kenya. The descriptive research design was utilized to assess the current status of community knowledge, perceptions, and attitudes toward wetland ecosystems and the environmental impacts associated with different land use practices. According to Mugenda and Mugenda (2009), descriptive design is essential for collecting data that reflect a population's opinions, behaviours, and experiences regarding a specific phenomenon of interest. In this context, the design enabled the collecting and analysis of qualitative and quantitative data on community awareness

and perceptions, providing insight into how local understanding might influence conservation practices and land use decisions.

In parallel, an experimental research design was incorporated to examine potential causal relationships between catchment land use practices such as agriculture, settlement expansion, deforestation, and changes in key water quality parameters. As described by Kothari (2004), experimental research allows researchers to establish cause-and-effect relationships through controlled observation and analysis. Water samples were collected from different sites within the river basins that exhibited varying degrees of land use intensity, and these samples were analyzed for specific indicators such as pH, turbidity, biological oxygen demand (BOD), dissolved oxygen (DO), and nutrient levels. Together, these complementary research designs enabled a robust exploration of the study's perceptual and empirical dimensions, facilitating a deeper understanding of how catchment land use practices affect the ecological health of small riverine wetland ecosystems in the study area.

3.4 Sampling Design

3.4.1 Survey Sampling Design

Farmers residing within the wetland catchment areas of Maragua and Mathioya River Basins were purposively recruited into the study based on three key criteria: willingness to participate, availability, and literacy level. A total of 30 farmers were selected as respondents. This sample size was deemed sufficient in line with (Cohen *et al.*, 2017), who suggest that a minimum of 30 participants can capture a broad range of opinions within a population and permit fundamental statistical analysis, particularly in exploratory or small-scale studies.

Structured questionnaires were administered to the selected respondents to gather primary data. The questionnaire was designed to assess farmers' knowledge and attitudes regarding the impacts of catchment land use on the water quality of small riverine wetlands. Two or more items represented each of the two primary constructs of knowledge and attitude to ensure internal consistency and improve measurement robustness.

Respondents' opinions were captured using a nominal scale, where participants indicated whether they agreed or disagreed with various statements related to land use practices and their perceived environmental impacts. Before full-scale data collection, a pilot study was conducted to assess the reliability of the questionnaire items. Cronbach's Alpha coefficient was used as the reliability index to evaluate the internal consistency of the items. A high Cronbach's Alpha value (typically ≥ 0.7) would confirm that the questionnaire items reliably measured the intended constructs. The sample questionnaire used in this study is provided in Appendix I.

3.4.2 Experimental Sampling Design

Standardized water and sediment sampling procedures were employed to ensure the collected data's consistency, reliability, and validity. Twenty four water and sediment samples were collected concurrently from six designated sampling points during dry and rainy seasons to account for seasonal variability in pollutant loads and ecosystem response. The six sampling locations were strategically selected to represent the upstream, midstream, and downstream sections of the two small riverine wetland ecosystems, Maragua and Mathioya Rivers, in Murang'a County. Ichichi (36.8441°) and Nyagatugu (36.86323° (36° 51' 48" E), were the upstream sampling points, Gachocho ($\approx -0.78^\circ$, 37.01° (elevation $\sim 1,600\text{--}1,700$ m) and Iyiego (37.03333° (37° 2' E), the midstream, and Mbombo (0° 47' 45.7" S, 37° 7' 58.5" E) and Gikuu ($\sim 0^\circ 34'\text{--}1^\circ 7'$ S, $36^\circ\text{--}37^\circ 27'$ E), were the downstream sampling points. This spatial distribution allowed for a comparative analysis of water quality and sediment characteristics along the longitudinal gradient of each river, thereby capturing the influence of varied land use practices and potential pollution sources across different segments of the catchment.

3.4.3 Water Sampling and Preservation

A total of 24 water samples were collected during both the dry (January-March) and wet (August-December) seasons across the wetland, that is, from up down stream. Water sample collection and analysis followed standardized protocols (Schmidt, 2003) to ensure the integrity and accuracy of results. One-litre water samples were collected using the grab sampling technique, carefully packed in clean containers, stored in cool boxes, and transported promptly to the laboratory for analysis. For nutrient analysis, 500 ml water samples were collected in plastic containers and taken to the lab where they were first treated prior analysis to prevent degradation of nutrient content. Samples intended for pesticide analysis were stored at a controlled temperature of 0–4°C before laboratory testing to preserve chemical stability. In situ measurements of physico-chemical parameters, including pH, salinity, electrical conductivity (EC), turbidity, and total dissolved solids (TDS), were conducted at the sampling sites using a hand-held portable water quality monitoring meter. To preserve the collected samples, 10% of a kilogram of sodium chloride (NaCl) was added, following procedures recommended by Akerblom (1995).

3.4.4 Sediment Sampling and Preservation

Twenty-four sediment samples were systematically collected from various riverbank locations, the upstream, mid- and Lowerstream, during the dry and rainy seasons to capture seasonal variations in sediment quality. At each designated sampling site along the riverine wetlands, two field points were randomly selected within the sediment zone at a 0–20 cm depth interval. This particular depth range was chosen because it is widely recognized as the most susceptible to contamination due to its proximity to surface activities and critical role in sediment-water interactions. According to Nowell *et al.* (1999), sediments within this shallow zone typically exhibit the highest concentrations of pollutants and have the most extraordinary capacity for ion exchange processes with the overlying water column, thus directly influencing water chemistry and aquatic ecosystem health.

To ensure the integrity of the samples, sediment was carefully collected using a stainless steel spoon, which minimizes contamination from sampling equipment. Each sample was then promptly wrapped in aluminum foil to protect it from light exposure and oxidation, which could alter the chemical composition. After wrapping, samples were sealed in airtight plastic bags, clearly labeled with relevant site and date information, and stored in ice boxes to maintain low temperatures during transportation. This cold storage helps to retard biological and chemical changes before laboratory processing.

Once at the laboratory, the sediment samples were stored at a controlled temperature range of 0–4°C to preserve their original state until further processing. Before chemical extraction and instrumental analysis, the samples underwent preparation procedures designed to isolate the contaminants of interest, ensuring a reliable and accurate assessment of sediment quality and pollutant concentrations.

3.5 Laboratory Analytical Techniques

3.5.1 Nutrients Analysis

Nutrient concentrations in the water samples were analyzed following the standard analytical methods recommended by APHA/AWWA/WEF (2005) to ensure accuracy and comparability of results. The key nutrients analyzed included Ammonia-nitrogen ($\text{NH}_3\text{-N}$), Nitrate-nitrogen ($\text{NO}_3^-\text{-N}$), and Phosphates (PO_4^{3-}), all of which are critical indicators of water quality and potential pollution sources.

Nitrate-nitrogen levels were determined using the Cadmium Reduction method, a widely accepted technique for nitrate quantification. In this method, nitrate ions (NO_3^-) present in the sample are chemically reduced to nitrite ions (NO_2^-) by passing the sample through a column containing cadmium granules (Alahi *et al.*, 2018). These cadmium granules are pre-treated with copper(II) sulfate to create a copper-coated surface, enhancing nitrate reduction efficiency. Following reduction, the amount of nitrite formed is quantified by a colorimetric reaction: nitrite reacts with sulphanilamide to form a diazonium salt, which then couples with N-(1-naphthyl) ethylenediamine to produce a brightly colored azo dye. The intensity of this dye correlates with the nitrite concentration and is measured using a spectrophotometer. Parallel analysis is conducted without the cadmium reduction step to accurately determine any nitrite originally present in the sample before reduction. This allows the total nitrate concentration correction by subtracting the native nitrite concentration (Droic & Vrtovsek, 2010).

Ammonia-nitrogen concentration was measured using a colorimetric method involving Nessler's reagent (K_2HgI_4) (Li *et al.*, 2011). In this reaction, ammonia in the water sample reacts with Nessler's reagent to produce a yellow-to-brown colloidal complex, the intensity of which reflects the ammonia concentration. Before this, water samples were acidified with sulfuric acid (H_2SO_4) to preserve the ammonia and prevent its loss as gas. They were then treated with zinc sulfate (ZnSO_4) and sodium hydroxide (NaOH) to precipitate interfering substances and create the ideal chemical environment for the Nessler reaction.

Phosphate concentrations were quantified using ascorbic acid, a sensitive and reliable colorimetric technique (Habibah *et al.*, 2018). In this process, orthophosphate ions (PO_4^{3-}) react with ammonium molybdate and potassium antimony tartrate in an acidic medium to form a yellow heteropoly-phosphomolybdic acid complex. Ascorbic acid reduces this complex to create an intensely blue-colored compound known as molybdenum blue. The color intensity, directly proportional to the phosphate concentration, was measured spectrophotometrically using an infrared phototube. This method allows precise quantification even at low phosphate levels, which is critical for assessing nutrient loading in wetland waters.

3.5.2 Extraction and Clean-up of Samples for Pesticide Analysis

3.5.2.1 Extraction and Clean-up of Sediment Samples

Approximately 20 grams of sediment samples were thoroughly ground with 30 grams of anhydrous sodium sulfate in a mortar. Additional sodium sulfate was gradually added during grinding to help absorb moisture from the sample, thereby improving extraction efficiency. The resulting dry mixture was then carefully transferred into an extraction column. The sample was eluted with 120 ml of dichloromethane, while gently shaking the column to enhance solvent contact and extraction.

After elution, the solvent extract was allowed to settle undisturbed for about fifteen minutes to enable any suspended particles or residual sodium sulfate to separate. The clear liquid was then carefully decanted into an evaporation flask. To ensure maximum recovery of the extract, the remnants of sodium sulfate left in the column after decanting were rinsed with an additional 15 to 30 ml of dichloromethane, and these rinses were combined with the initial extract. The combined extract fractions were filtered through a glass wool plug to remove particulate impurities. The filtered extract was then concentrated using a rotary evaporator. Once concentrated to dryness, the residue was reconstituted in 2 ml of cyclohexane, which served as the solvent for subsequent purification steps (Akerblom, 1995).

For further purification, the extract underwent clean-up by column chromatography. The chromatographic column was packed sequentially with magnesium silicate (Florisil) as the adsorbent, anhydrous sodium sulfate to remove residual moisture, and a plug of glass wool to hold the packing in place and prevent clogging. Before loading the sample, the column was conditioned by passing 50 ml of cyclohexane dropwise, allowing the solvent to wet the adsorbent evenly and prepare the column for efficient separation.

Following conditioning, the 2 ml concentrated sample was carefully loaded onto the top of the column and eluted through the adsorbent. The eluate was collected and concentrated back down to 2 ml, ensuring the removal of impurities while retaining the target analytes. Finally, the purified extract was transferred into a Teflon-capped vial and stored at 0–4°C to preserve sample integrity before instrumental analysis.

3.5.2.2 Water Samples Extraction and Clean-up

A one-liter unfiltered water sample, preserved by adding 10% sodium chloride, was subjected to extraction using the Liquid-Liquid Extraction (LLE) method following the procedure outlined by Akerblom (1995). The entire one-liter sample was transferred into a one-liter separating funnel for extraction. Dichloromethane, an organic solvent immiscible with water, was added to the funnel to selectively extract the target analytes from the aqueous phase into the organic phase.

After thorough mixing and phase separation, the extracted compounds' organic layer was collected and filtered through a glass wool plug embedded with approximately 30 grams of anhydrous sodium sulfate to remove residual water and dry the extract efficiently. To ensure maximum recovery, the sodium sulfate plug was rinsed with additional dichloromethane, and these rinses were combined with the initial filtrate.

The combined extract was then concentrated under reduced pressure (in vacuo) at a controlled temperature of 30°C to gently evaporate the solvent without degrading the analytes. Cyclohexane was used as the solvent during this concentration process to facilitate reconstitution and to improve compatibility with subsequent analytical instrumentation. Finally, the concentrated extract was reconstituted into 2 ml vials, ready for instrumental analysis using HPLC-UV.

To generate chromatograms, the environmental samples and reference pesticide standards were analyzed under identical instrumental conditions. Identification of pesticides in the samples was achieved by comparing the retention times and mass spectral data of sample peaks with known reference standards. Quantification was performed by measuring the peak areas of the pesticide analytes detected in the samples and comparing these to the peak areas of corresponding standards with known concentrations. This comparative approach enabled accurate determination of pesticide concentrations within the samples, ensuring precise and reliable analytical results (Kipkemai *et al.*, 2020; Martens *et al.*, 1999).

3.5.3 Method validation for water and sediments pesticide residues using HPLC-UV analysis.

3.5.3.1 Standard solutions and standard curve plotting

Stock standard solutions of each pesticide were initially prepared at a concentration of 1000 ppm using acetonitrile as the solvent. These concentrated stock solutions created working standards of 1 ppm through appropriate dilution. To establish a calibration curve for each pesticide, a series of diluted solutions were prepared by further diluting the working standards with the mobile phase, which served as the diluent to ensure compatibility with the HPLC system. The range of concentrations varied depending on the pesticide: for thiamethoxam, concentrations of 0.01, 0.02, 0.03, 0.041, and 0.051 ppm were prepared; for diazinon, the concentrations included 0.01036, 0.02072, 0.03018, 0.04414, 0.0518, 700.0, and 1000.0 ppm; for malathion, concentrations of 0.008, 0.016, 0.024, 0.032, and 0.040 ppm were prepared; while for carbendazim, concentrations of 0.01004, 0.02008, 0.03012, 0.04016, and 0.0502 ppm were used. Each solution was then individually injected into the High-Performance Liquid Chromatography (HPLC) system, and the resulting peak areas were carefully recorded. These peak areas were plotted against their corresponding concentrations to generate standard calibration curves. These were subsequently used to quantify the pesticide residues in environmental samples by relating their chromatographic responses to known concentrations.

3.5.3.2 Percentage recovery

The accuracy of the analytical method used for determining the three pesticides was evaluated through a percentage recovery study. This was done by spiking distilled water, confirmed to be free of any pesticide residues, with known concentrations of each pesticide at three different levels: 0.008 ppm, 0.024 ppm, and 0.060 ppm. These spiked samples underwent the same extraction procedures as the actual water and sediment samples to ensure consistency and comparability. Subsequently, the extracted samples were analyzed using the same High-Performance Liquid Chromatography with Ultraviolet detection (HPLC-UV) method. The recovery results obtained from these analyses were within the acceptable range of 70% to 120%, indicating that the technique was accurate and reliable for detecting and quantifying pesticide residues in the environmental samples, consistent with established guidelines (Hill, 2000).

3.5.3.3 Precision of the method

The precision of the analytical method was assessed by calculating the relative standard deviation (RSD) of the peak areas obtained from six replicate injections of each pesticide at three concentration levels: 0.008 ppm, 0.024 ppm, and 0.060 ppm. This approach allowed for evaluating the method's repeatability and consistency in producing reliable results under the same operating conditions. Analyzing the variation in peak areas across multiple injections, the RSD provided a quantitative measure of the method's precision, ensuring that the analytical procedure yields reproducible data necessary for accurate pesticide quantification.

3.6 Metal Ion Detection

Metal ion analysis was done using Agilent 4210 MP-AES instrument. Systematic stages of analysis began with sample preparation where water samples were typically digested using acid digestion and the resulting solution was diluted and filtered to ensure compatibility with the instrument; this was then followed by instrument calibration where Calibration curves were created using standards with known concentrations of target metal ions; the standards were used to plot intensity vs. concentration; the prepared sample was then aspirated into a nebulizer, which converted it into a fine mist, where the high energy causes metal atoms to emit light; detection then happened where emission from excited atoms was collected at element-specific wavelengths (e.g., 324.75 nm for Cu); The intensity of the emission is directly proportional to the concentration of the metal ion. Finally, quantification was done where the instrument software compared the emission intensity from the sample to the calibration curve to determine the concentration of each metal ion.

3.7 Water Quality Parameters Analysis

Water physico-chemical parameters, that is, salinity, total dissolved solids (TDS), electrical conductivity (EC), turbidity, and p^H were measured using hand-held portable water quality monitor meter (Shin *et al.*,2018).

Specific quality control (QC) procedures and calibration frequencies were followed in each parameter analysis to ascertain accuracy, reliability of data and precision of the analytical results. These quality protocols help detect chances or levels of contamination, equipment drift, as well as procedural errors. Both sample duplicates and replicate analyses were performed. Sample duplicates: helped in assessing

precision of the analysis where differences between duplicates would indicate variability. Replicate Analyses: Involved multiple measurements on the same sample to assess method repeatability. Also, use of blanks, maintenance and cleaning of instruments were observed to ensure quality control. Regular calibration of the instruments was done using specific certified standards for each analysis.

3.8 Data Analysis

Data preprocessing was the first critical step in ensuring the accuracy and reliability of the research findings. This involved thoroughly checking the dataset for missing values, inconsistencies, or outliers that could distort the analysis. By carefully cleaning the data, the study maintained its relevance and validity concerning the research objectives. After this, all variables were systematically coded to facilitate efficient data entry and management. The coded data were then organized into a spreadsheet format, aligning each variable with the corresponding water quality measurements for further analysis.

The statistical analysis was conducted using the Statistical Package for the Social Sciences (SPSS) software, employing descriptive and inferential methods. Descriptive statistics were used to provide an overview of the data through frequency distributions and percentages, helping to summarize the characteristics of the sample and the measured parameters. Additionally, error bars were utilized in graphical representations to illustrate variability and confidence intervals, which helped visually communicate the measurements' precision across different sampling points and seasons.

For inferential statistics, Pearson correlation analysis was conducted to explore and quantify the relationships between various water quality parameters. This helped me understand how one variable might influence or relate to another within the studied ecosystems. Analysis of variance (ANOVA) was then performed to test for significant differences among groups, particularly focusing on seasonal variations and differences between catchment areas. Upon finding substantial effects with ANOVA, post-hoc tests were necessary to pinpoint precisely which groups differed. The Least Significant Difference (LSD) test was used as the post-hoc method, enabling multiple pairwise comparisons between group means. This approach provided detailed insights into

specific differences in water quality indicators during the dry and rainy seasons and across different land use types within the wetland catchments.

To assess the **Pollution Index (PI)** for each heavy metal, the following formula was used:

$$\text{Pollution Index (PI)} = C_i / S_i$$

Where: C_i - is the measured concentration of metal (mg/L),

S_i - is the **standard permissible limit** (mg/L) of the metal in water (from WHO or national environmental standards).

[Pollution index categories: $PI \leq 1$: Low Pollution; $1 < PI \leq 5$: Moderate Pollution and $PI > 5$: High Pollution]

3.9 Ethical Issues

The researcher first obtained the authorization to conduct the study by securing a letter of authority from the National Commission for Science, Technology and Innovation (NACOSTI). A copy of this authorization letter was subsequently submitted to the County Commissioner of Murang'a County as part of the formal protocol to ensure compliance with local regulations. Before data collection, the respondents were fully informed about the objectives and purpose of the research. They were briefed on their rights, including the voluntary nature of participation and the freedom to withdraw from the study without any negative consequences. To build trust and encourage honest responses, the researcher assured participants that all information provided would be kept strictly confidential and used solely for research purposes.

3.10 Summary

This chapter has provided a comprehensive overview of the materials and methods employed throughout the research to ensure the achievement of the study's objectives. It began by describing the study area, highlighting key geographical and environmental features relevant to the investigation. The chapter then detailed the sampling designs, including the selection criteria and procedures for water and sediment samples. Further, it outlined the sample preparation steps and the various analytical techniques for measuring the different parameters of interest. Additionally,

the chapter explained the procedures implemented to maintain Analytical Quality Assurance for both water and sediment samples, ensuring the reliability and accuracy of the data collected. Finally, it described the data analysis methods, including descriptive and inferential statistical techniques, and pollution index (PI) analyses to interpret the results and draw meaningful conclusions.

CHAPTER FOUR

RESULTS AND DISCUSSION

4.1 Introduction

This chapter presents the results and discusses the research findings conducted in Mathioya and Maragua wetland ecosystems in Murang'a County, Kenya. It provides a report on the impacts of catchment land use practices on the water quality of small wetland ecosystems in Murang'a County. It also establishes farmer's knowledge and attitude towards wetland conservation. This chapter also reports on agricultural practices carried out along the wetland ecosystems, which involve various agrochemicals, the residue levels of nutrients and pesticide inflows in water and sediment samples and effects of land use practices on water quality on small wetland ecosystems.

4.2 Status of Farmers' Knowledge and Attitudes on the effects of catchment land use on wetland water quality in Murang'a County, Kenya

The relationship between catchment land use and wetland water quality is critically important, particularly in agricultural regions where land management practices can directly impact the health of wetland ecosystems. In Murang'a County, Kenya, where agriculture plays a significant role in the local economy, understanding farmers' knowledge and attitudes regarding the effects of catchment land use on wetland water quality is essential for sustainable resource management. This analysis aims to establish the status of farmers' knowledge and attitudes concerning the effects of catchment land use on wetland water quality in Murang'a County. By examining correlations between statements related to wetland conservation, agricultural practices, and environmental concerns, we seek to uncover insights into farmers' perceptions, awareness levels, and behaviors regarding wetland protection and water quality management.

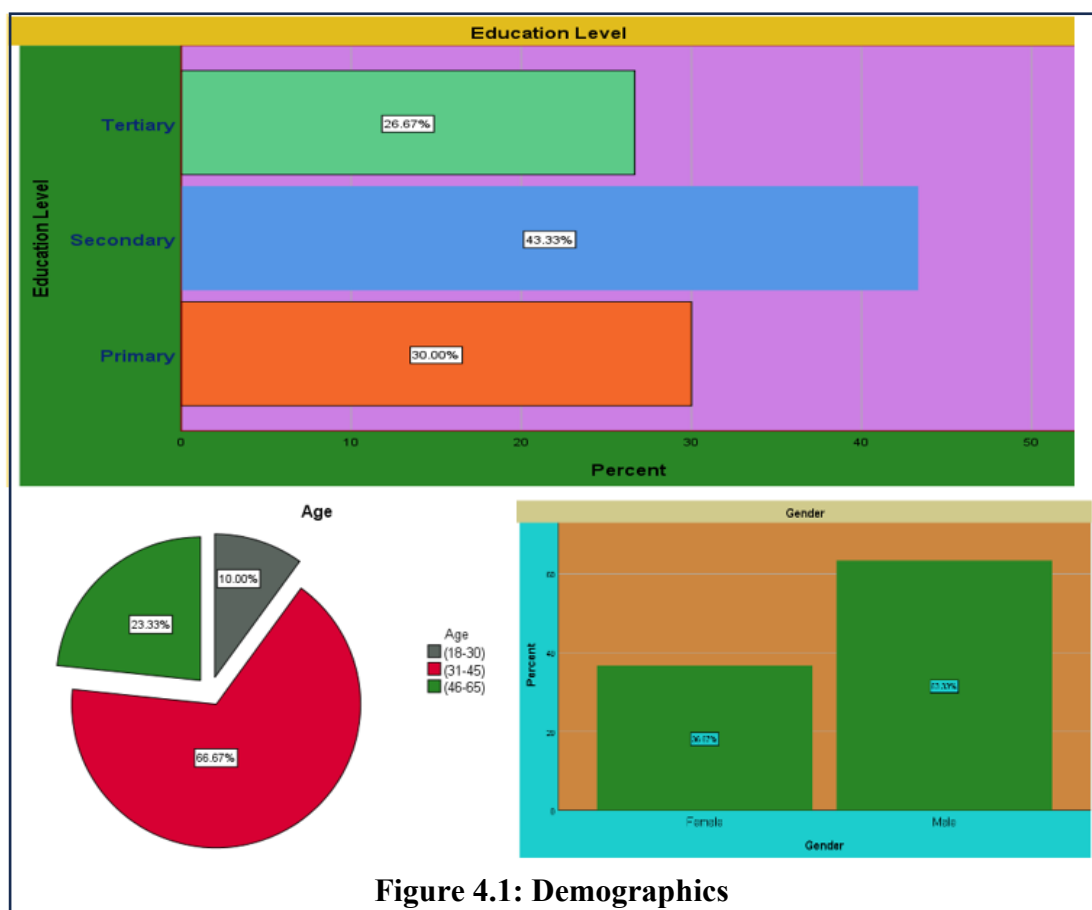
Descriptive statistics are like a data snapshot, capturing its essence simply and understandably. They help us explore and understand data by giving us a quick overview of its key features. By summarizing things like averages, spreads, and patterns, descriptive statistics make it easier to compare data, spot trends, and make informed decisions. They are not just numbers; they are insights that guide research, communication, and decision-making processes. Whether identifying outliers,

cleaning up messy data, or generating new ideas, descriptive statistics are the starting point for unlocking the stories hidden within the numbers. In this analysis, descriptive statistics is structured into demographics analysis and farmers' knowledge and attitude analysis.

4.2.1 Demographics Analysis

The survey data presents the gender distribution of respondents, with 36.7% identifying as female and 63.3% as male. This reveals a higher representation of males within the surveyed population. On the age issue, the survey indicates that most respondents fall within the (31-45) age group, constituting 66.7% of the total respondents. The (46-65) age group comprises 23.3% of the respondents. The (18-30) age group represents the smallest portion, with only 10.0% of respondents falling into this category. Therefore, the survey sample predominantly comprises individuals aged between 31 and 45 years. The survey has a smaller representation of younger adults (18-30) and older adults (46-65).

The survey data reveals the educational backgrounds of respondents, showcasing a diverse mix of primary, secondary, and tertiary education levels. Primary education is represented by 30.0% of respondents, indicating a significant portion with foundational educational attainment. Secondary education is the most prevalent category, encompassing 43.3% of respondents, signifying a substantial portion with completed high school or equivalent education. Tertiary education, comprising higher education like college or university, represents 26.7% of respondents, indicating a notable proportion with advanced educational achievements (Figure 4.1). Also, survey revealed that most farmers at mid and Lowerstream points conduct vegetable and fruit farming practices, while a few in the upstream points of riverine wetlands conduct tea farming. They also confirmed their application of synthetic fertilizers for crop yield concern and mostly using organophosphate class of pesticide for crop protection during pest management.



4.2.2 Farmers' Knowledge and Attitude Analysis

The analysis of farmers' knowledge and attitudes involved surveying a representative sample to understand their comprehension and perceptions regarding agricultural practices on catchment ecosystems, innovations, and broader issues. Through careful survey design and data collection, key insights into farmers' understanding of farming techniques and their attitudes toward sustainability, government policies, and market dynamics were gleaned. Statistical analysis helped identify demographic disparities and correlations, while interpretation of findings leads to actionable recommendations for policymakers and stakeholders.

The data indicates that most respondents, accounting for 83.3%, can identify wetlands in their neighborhood, while 16.7% cannot. This suggests a significant awareness and recognition of wetland areas among the surveyed population. From these, most respondents, comprising 90.0%, recognize the importance of wetlands to them directly, while 10.0% indicate uncertainty regarding this significance (Figure 4.2). This high acknowledgment suggests that wetlands hold tangible value for the surveyed

population, likely due to their ecological, economic, and cultural benefits. Additionally, a vast majority of respondents, accounting for 93.3%, recognize the importance of wetlands to all of us, while only 6.7% indicate otherwise. This high level of agreement underscores the widespread understanding among the surveyed population regarding the significance of wetlands for society. Among the respondents, 53.3% believe that the size of the wetland around them has decreased, while 26.7% perceive no change, and 20.0% are uncertain. This suggests a prevailing concern among a significant portion of the surveyed population regarding the decline in the size of wetlands.

1: I can identify wetlands in my neighborhood					
		Frequency	Percent	Valid Percent	Cumulative Percent
Valid	No	5	16.7	16.7	16.7
	Yes	25	83.3	83.3	100.0
	Total	30	100.0	100.0	

2: Wetlands are important to me directly					
		Frequency	Percent	Valid Percent	Cumulative Percent
Valid	I do not know	3	10.0	10.0	10.0
	Yes	27	90.0	90.0	100.0
	Total	30	100.0	100.0	

3: Wetlands are important to all of us					
		Frequency	Percent	Valid Percent	Cumulative Percent
Valid	No	2	6.7	6.7	6.7
	Yes	28	93.3	93.3	100.0
	Total	30	100.0	100.0	

Figure 4.2: Wetlands and its importance

Regarding whether wetlands drain into the River Mathioya, the data indicates that the majority of respondents, comprising 73.3%, believe that wetlands do indeed drain into the River Mathioya. In comparison, 23.3% think otherwise, and 3.3% are uncertain (Figure 4.2). This suggests a strong perception among the surveyed population regarding the connectivity between wetlands and the river, highlighting awareness of hydrological interactions and potential impacts on water quality and ecosystem health.

Regarding the impact of tree cutting on wetlands, the data reveals that 46.7% of respondents believe that cutting trees has negatively impacted wetlands, while 10.0% perceive no such impact, and 43.3% are uncertain (Figure 4.3).

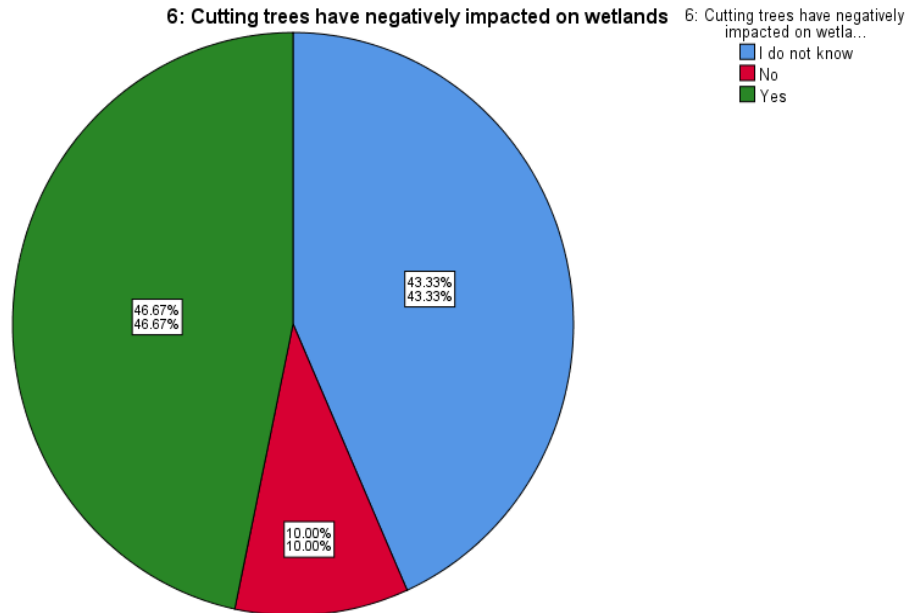


Figure 4.3: Impacts of cutting trees on wetlands

This underscores a significant level of uncertainty among respondents regarding the relationship between tree cutting and wetland health, potentially indicating a need for further education and awareness-raising on the ecological importance of trees for wetland ecosystems.

Lastly, regarding the potential harm from agrochemical usage on wetlands, the data shows that 23.3% of respondents believe the amounts of agrochemicals used can harm wetlands. In comparison, 20.0% think otherwise, and 56.7% are uncertain (Figure 4.4). This points to a prevalent lack of awareness or uncertainty regarding the potential impacts of agrochemicals on wetland ecosystems, suggesting a need for increased education and information dissemination on sustainable agricultural practices and their implications for wetland conservation.

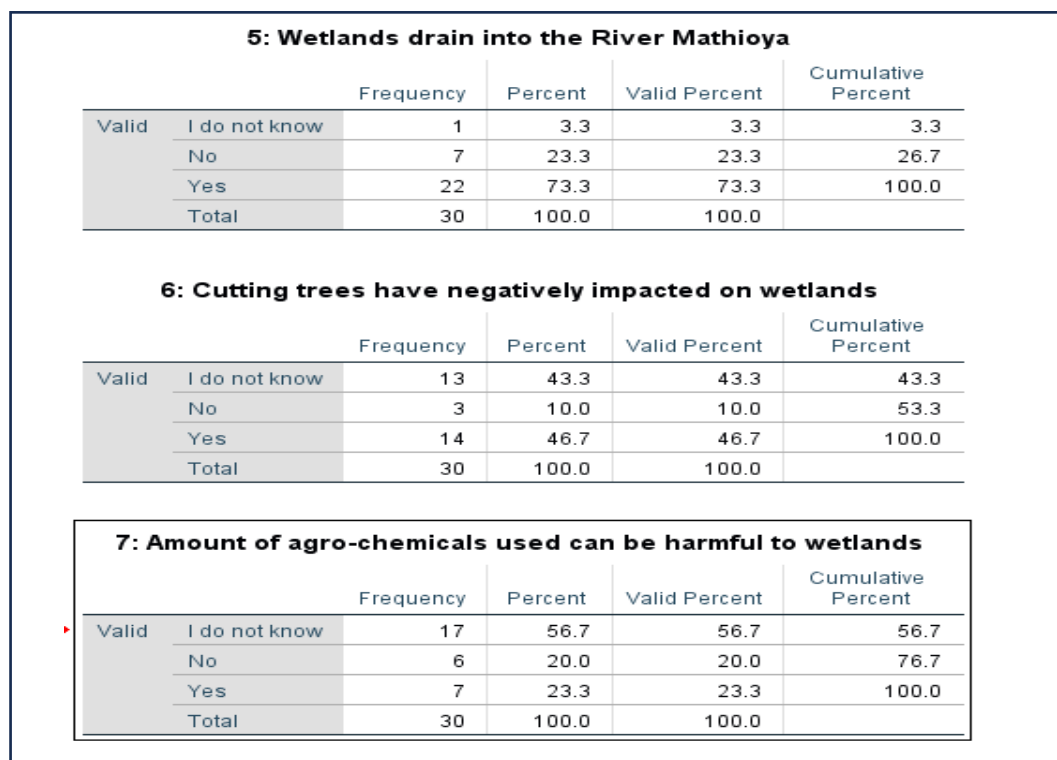


Figure 4.4: Drainage and implication on wetlands

On whether the type of agrochemicals used can harm wetlands, the results indicated that most respondents, comprising 80.0%, believe that certain agrochemicals can harm wetlands. In comparison, 20.0% think otherwise (Figure 4.5). This underscores a widespread perception among the surveyed population regarding the potential negative impacts of agrochemicals on wetland ecosystems, highlighting the need for environmentally responsible agricultural practices.

Also, about the method of applying agrochemicals and its potential harm to wetlands, Table 3 reveals that 33.3% of respondents believe that applying agrochemicals can harm wetlands. In comparison, 13.3% perceive no such harm, and 53.3% are uncertain. This shows a significant level of uncertainty among respondents regarding the potential impacts of agrochemical application methods on wetland health. This suggests a need for further education and awareness-raising on best agricultural practices for pesticide and fertilizer application to minimize the environmental effects.

Concerning farmers' awareness of safe agricultural practices, the data shows that 70.0% of respondents believe that farmers are not fully aware of safe farming

practices, while only 30.0% think otherwise (Table 4.3). This highlights a prevalent perception among the surveyed population regarding the need for increased education and training on sustainable and environmentally friendly farming practices to mitigate adverse impacts on wetlands and other ecosystems important to humans and aquatic organisms.

8. Type of agrochemicals used can be harmful to wetlands					
		Frequency	Percent	Valid Percent	Cumulative Percent
Valid	No	6	20.0	20.0	20.0
	Yes	24	80.0	80.0	100.0
	Total	30	100.0	100.0	

9. The method of applying agrochemicals can be harmful to wetlands					
		Frequency	Percent	Valid Percent	Cumulative Percent
Valid	I do not know	16	53.3	53.3	53.3
	No	4	13.3	13.3	66.7
	Yes	10	33.3	33.3	100.0
	Total	30	100.0	100.0	

10. Farmers are aware of safe agricultural practices					
		Frequency	Percent	Valid Percent	Cumulative Percent
Valid	No	21	70.0	70.0	70.0
	Yes	9	30.0	30.0	100.0
	Total	30	100.0	100.0	

Figure 4.5: Agrochemicals and practices by farmers

The data discloses that a majority of respondents, comprising 83.3%, believe that farmers hardly care about the existence of wetlands, while 16.7% think otherwise (Figure 4.6).

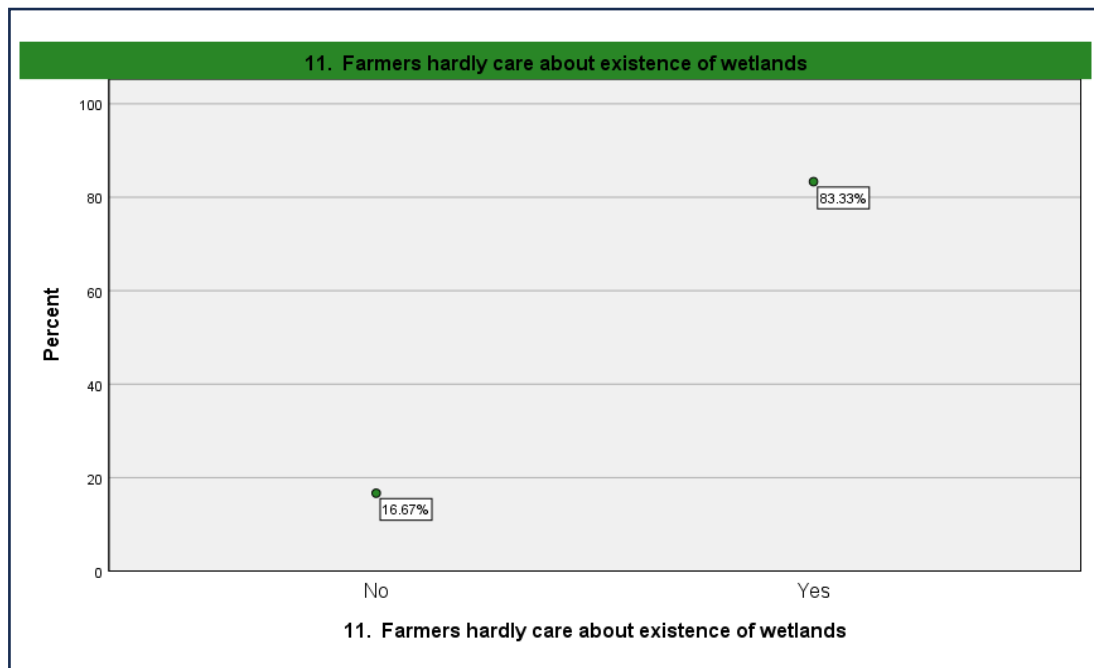


Figure 4.6: Conservation of existing wetlands

Equally, regarding farmers' attitudes towards the type of agrochemicals they use on their farms, 83.3% of respondents perceive that farmers hardly care about the kind of agrochemicals they use. Finally, regarding farmers' concerns about wetland conservation, 76.7% of respondents believe that farmers are hardly concerned about wetland conservation, while 23.3% think otherwise (Figure 4.7).

11. Farmers hardly care about existence of wetlands					
		Frequency	Percent	Valid Percent	Cumulative Percent
Valid	No	5	16.7	16.7	16.7
	Yes	25	83.3	83.3	100.0
	Total	30	100.0	100.0	

12 Farmers hardly care about the type of agro chemicals they use in their farms					
		Frequency	Percent	Valid Percent	Cumulative Percent
Valid	No	5	16.7	16.7	16.7
	Yes	25	83.3	83.3	100.0
	Total	30	100.0	100.0	

13. Farmers are hardly concerned about conservation of wetlands					
		Frequency	Percent	Valid Percent	Cumulative Percent
Valid	No	7	23.3	23.3	23.3
	Yes	23	76.7	76.7	100.0
	Total	30	100.0	100.0	

Figure 4.7: Conservation and environmental practices

The statistics reveal that a majority of respondents, comprising 76.7%, believe that farmers are hardly concerned about the safety of the methods they use in applying agrochemicals, while 23.3% think otherwise (Figure 4.8).

14. Farmers are hardly concerned about safety of methods they use in applying agro					
		Frequency	Percent	Valid Percent	Cumulative Percent
Valid	No	7	23.3	23.3	23.3
	Yes	23	76.7	76.7	100.0
	Total	30	100.0	100.0	

15 Farmers hardly find value in attending conservation awareness forums					
		Frequency	Percent	Valid Percent	Cumulative Percent
Valid	No	5	16.7	16.7	16.7
	Yes	25	83.3	83.3	100.0
	Total	30	100.0	100.0	

Figure 4.8: Awareness and safety concerns on wetlands

Concerning farmers' participation in conservation awareness forums, 83.3% of respondents perceive that farmers hardly find value in attending such forums, while 16.7% think otherwise (Figure 4.9).

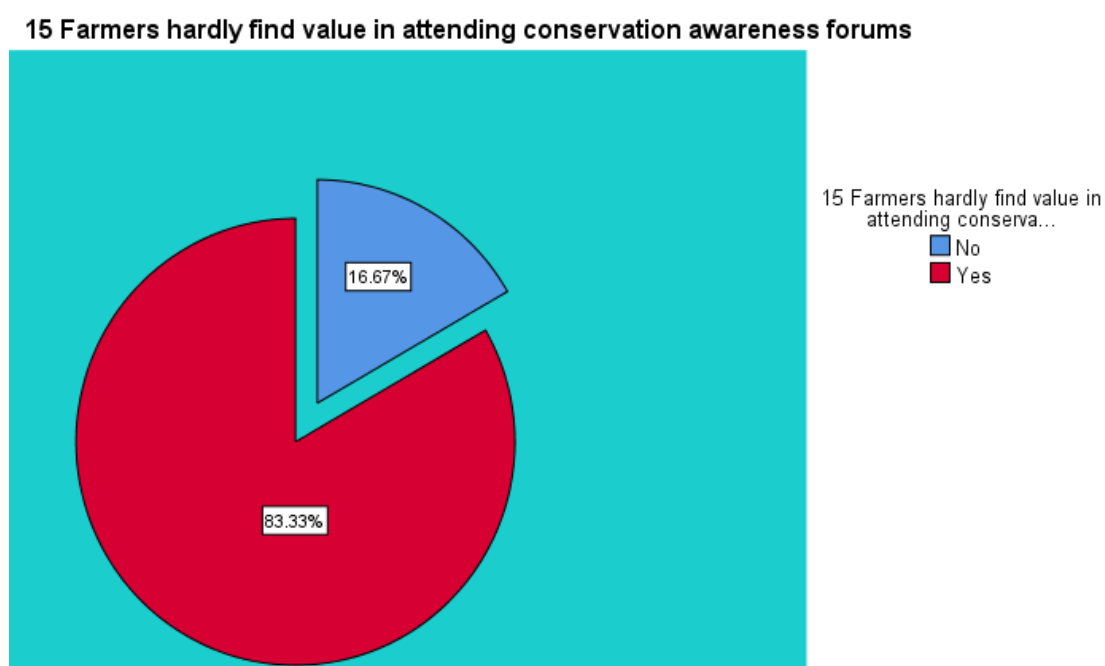


Figure 4.9: Attending awareness forums

Lastly, regarding the application of conservation methods by farmers, 73.3% of respondents believe that farmers hardly apply the conservation methods they are aware of, while 26.7% think otherwise (Figure 4.10).

According to the data, 73.3% of respondents believe that farmers hardly find the need to observe safety regulations when applying chemicals in farming, while 26.7% think otherwise. Additionally, regarding farmers' concern about the effects of agrochemicals they use on the farm, 70.0% of respondents believe that farmers hardly care about these effects, while 30.0% think otherwise.

16. Farmers hardly apply conservation methods they are aware of					
		Frequency	Percent	Valid Percent	Cumulative Percent
Valid	No	8	26.7	26.7	26.7
	Yes	22	73.3	73.3	100.0
	Total	30	100.0	100.0	

17. Farmers hardly find the need to observe safety regulations when applying chemicals in					
		Frequency	Percent	Valid Percent	Cumulative Percent
Valid	No	8	26.7	26.7	26.7
	Yes	22	73.3	73.3	100.0
	Total	30	100.0	100.0	

18. Farmers hardly care about the effects of agro-chemicals they use in the farm on					
		Frequency	Percent	Valid Percent	Cumulative Percent
Valid	No	9	30.0	30.0	30.0
	Yes	21	70.0	70.0	100.0
	Total	30	100.0	100.0	

Figure 4.10: Application of conservation methods

The figures show that a majority of respondents, accounting for 76.7%, believe that farmers are more concerned with the yields they get from the farm than with safety considerations. Conversely, 23.3% of respondents disagree with this perception (Figure 4.11). Similarly, regarding farmers' concerns about the pollution of wetlands, 80.0% of respondents believe that pollution of wetlands is not a concern for farmers, while 20.0% think otherwise.

19. Farmers are more concerned with the yields they get from the farm than safety of					
		Frequency	Percent	Valid Percent	Cumulative Percent
Valid	No	7	23.3	23.3	23.3
	Yes	23	76.7	76.7	100.0
	Total	30	100.0	100.0	
20. Pollution of wetlands is not a concern for farmers					
		Frequency	Percent	Valid Percent	Cumulative Percent
Valid	No	6	20.0	20.0	20.0
	Yes	24	80.0	80.0	100.0
	Total	30	100.0	100.0	

Figure 4.11: Pollution of wetlands versus crop yields concerns

4.3 Analysis of farmers' knowledge and attitudes using Pearson correlations

Correlation analysis is essential for achieving the objective of establishing the status of farmers' knowledge and attitudes regarding the effects of catchment land use on wetland water quality in Murang'a County for several reasons: First, it allows us to identify relationships between different variables related to farmers' knowledge, attitudes, and behaviors. By examining correlations, we can understand how various factors, such as farmers' awareness levels and conservation practices, might be interrelated. Secondly, it helps in understanding patterns within the data. For example, we can determine if a positive correlation exists between farmers' awareness of wetland conservation and their participation in conservation activities. Similarly, we can explore whether certain attitudes or behaviors tend to co-occur among farmers. Third, they provide insights into potential predictors of certain attitudes or behaviors. For instance, if we find a strong positive correlation between farmers' education levels and their knowledge of wetland conservation, education may play a significant role in shaping awareness and behavior. Lastly, they can inform policy decisions and interventions by highlighting key factors influencing farmers' attitudes and behaviors related to wetland conservation.

The analysis of the Pearson correlation data provides an understanding of the factors influencing farmers' knowledge and attitudes regarding wetlands and agricultural practices. The knowledge-related questions (Q1-Q10) and attitude-related questions

(Q11-Q20) reveal significant correlations with age, education level, and gender and inter-correlations among the questions themselves as shown in Table 4.1.

i. Farmers' knowledge (Q1-Q10)

In the realm of knowledge, age appears to have a significant negative correlation with Q5 (-0.429 , $p = 0.018$), indicating that older farmers are less likely to be aware that wetlands drain into the River Mathioya. This suggests that younger farmers might be more informed about specific environmental features in their locality. Interestingly, no significant correlations were found between education level or gender and knowledge-related questions. This might indicate that formal education and gender do not play a crucial role in this aspect of environmental knowledge among the surveyed farmers.

Several significant inter-question correlations were observed within the knowledge questions. Particularly, there is a negative correlation between Q5 and Q7 (-0.382 , $p = 0.037$). This suggests that farmers who know that wetlands drain into the River Mathioya are less likely to believe that the amounts of agrochemicals used can harm wetlands. Similarly, Q5 also negatively correlates with Q9 (-0.393 , $p = 0.031$), and Q8 correlates negatively with Q9 (-0.385 , $p = 0.036$), highlighting inconsistencies or conflicts in farmers' knowledge about agrochemicals and their effects on wetland ecosystems.

ii. Farmers' attitudes (Q11-Q20)

When examining attitudes, education level shows a significant positive correlation with Q17 (0.374 , $p = 0.042$), indicating that higher education levels are associated with a greater likelihood of farmers recognizing the need to observe safety regulations when applying agrochemicals. This suggests that educational interventions could effectively improve attitudes towards safe agricultural practices. Gender differences are notable in attitudes towards wetlands conservation and pollution. Specifically, Q13 shows a significant negative correlation with gender (-0.420 , $p = 0.021$), indicating that men are less concerned about wetland conservation than women. Similarly, Q20 also has a significant negative correlation with gender (-0.380 , $p = 0.038$), suggesting that men are less concerned about the pollution of wetlands.

Several significant inter-correlations among the attitude questions further illustrate the complexity of farmers' attitudes. For example, Q9 correlates positively with Q12

(0.393, $p = 0.031$), suggesting that farmers who recognize the harmful methods of applying agrochemicals are likelier to believe that other farmers are indifferent to the type of agrochemicals used. Additionally, Q13 correlates positively with Q17 (0.380, $p = 0.038$), indicating that those less concerned about conservation are also less likely to observe safety regulations. Another notable correlation is between Q14 and Q18 (-0.361, $p = 0.05$), showing that in-difference towards safe application methods is linked to a lack of concern about the effects of agrochemicals on wetlands. The strongest correlation is between Q15 and Q18 (0.683, $p = 0.000$), suggesting a strong association between the lack of value placed on conservation awareness forums and indifference towards the impact of agrochemicals.

Based on the analysis, we reject the null hypothesis H_{01} . The evidence from the Pearson correlation analysis indicates that farmers in Murang'a County do not possess sufficient integrated knowledge about the effects of catchment land use on wetland water quality. The significant negative correlations between specific knowledge (e.g., wetland drainage) and understanding the impacts of agrochemical use suggest gaps in comprehensive environmental knowledge.

Table 4.1: Correlation analysis

Correlations												
		Age	Education Level	Gender	Q1	Q5	Q8	Q9	Q13	Q14	Q15	
Q5	P-Corr	-.429*	0.228	0.092	-0.085	1	0.032	0.014	-0.165	0.135	-0.085	
	Sig. (2-tailed)	0.018	0.227	0.629	0.655		0.868	0.942	0.384	0.477	0.655	
Q7	P-Corr	0.238	0.089	0.028	0.359	-.382*	0.201	-0.177	-0.032	-0.317	0.036	
	Sig. (2-tailed)	0.205	0.64	0.884	0.051	0.037	0.287	0.35	0.868	0.088	0.851	
Q9	P-Corr	0.052	-0.156	0.061	-.393*	0.014	-.385*	1	0.139	0.052	0.197	
	Sig. (2-tailed)	0.784	0.41	0.749	0.031	0.942	0.036		0.465	0.785	0.297	
Q12	P-Corr	-0.053	-0.139	0.031	0.04	0.255	-0.224	.393*	-0.035	-0.035	0.04	
	Sig. (2-tailed)	0.781	0.465	0.871	0.834	0.174	0.235	0.031	0.853	0.853	0.834	
Q13	P-Corr	-0.009	0.185	-.420*	0.176	-0.165	-0.079	0.139	1	0.255	-0.035	
	Sig. (2-tailed)	0.961	0.327	0.021	0.352	0.384	0.679	0.465		0.174	0.853	
Q17	P-Corr	-0.259	.374*	-0.302	0.135	-0.057	0.075	-0.05	.380*	0.202	0.135	
	Sig. (2-tailed)	0.166	0.042	0.104	0.477	0.764	0.692	0.794	0.038	0.284	0.477	
Q18	P-Corr	-0.233	-0.029	-0.196	-0.293	0.18	-0.145	0.256	-0.189	-.361*	.683**	
	Sig. (2-tailed)	0.215	0.879	0.299	0.116	0.342	0.443	0.172	0.317	0.05	0	
Q20	P-Corr	-0.178	-0.022	-.380*	0.224	0.032	0.167	-0.11	.512**	0.315	-0.224	
	Sig. (2-tailed)	0.347	0.907	0.038	0.235	0.868	0.379	0.563	0.004	0.09	0.235	
* Correlation is significant at the 0.05 level (2-tailed).												
** Correlation is significant at the 0.01 level (2-tailed).												

4.4 Analysis of Variance (ANOVA) on Farmers' Knowledge and Attitudes

iii. Differences based on education level

The ANOVA (Analysis of Variance) results help determine if there are statistically significant differences in the knowledge and attitudes of farmers in Murang'a County based on their education levels. In the knowledge questions (Q1-Q10), most show no significant differences across groups at both 5% and 10% significance levels, except for Q4 ($F(2,27) = 3.441, p = 0.047$), which is significant at the 5% level, indicating varying perceptions about wetland size decrease by education level. Q3 ($F(2,27) = 2.700, p = 0.085$) is significant at the 10% level but not at 5%, indicating the recognized importance of wetlands to the ecosystem. In the attitude questions (Q11-Q20), there are no significant differences at both 5% and 10% levels for most questions, except for Q16 ($F(2,27) = 3.241, p = 0.055$), which is significant at the 10% level, suggesting potential differences in the application of conservation methods. Q17 ($F(2,27) = 2.367, p = 0.113$) is close to significance at the 10% level.

These results suggest that while education level may not heavily influence most aspects of knowledge and attitudes, it can affect specific perceptions and practices related to wetland conservation strategies. This highlights the importance of targeted educational interventions to address gaps in farmers' understanding and application of conservation practices.

iv. Differences Based on Gender

In examining differences based on gender, most questions (Q1-Q20) show no significant differences at both the 5% and 10% significance levels. However, some exceptions exist. Q3 ("Wetlands are important to all of us") shows a potential difference at the 10% level ($p = 0.057$) but not at the 5% level. Q11 ("Farmers hardly care about the existence of wetlands") and Q15 ("Farmers hardly find value in attending conservation awareness forums") also approach significance at the 10% level ($p = 0.066$ for both) but not at the 5% level. Q13 ("Farmers are hardly concerned about conservation of wetlands") indicates a significant difference at the 5% level ($p = 0.021$). Similarly, Q20 ("Pollution of wetlands is not a concern for farmers") shows significance at the 5% level ($p = 0.038$). These findings suggest some gender-based differences in attitudes towards wetland importance and conservation.

v. Differences based on age

In investigative differences created by age, most questions show no significant differences at 5% and 10% significance levels, as indicated by p-values greater than 0.10. However, there are some notable exceptions. Q5 ("Wetlands drain into the River Mathioya") shows a significant difference at the 5% level ($p = 0.048$), indicating latent age-based differences in awareness or perception of this fact. None of the other questions show significant differences based on age, suggesting that age does not strongly influence perceptions or attitudes toward wetlands and conservation practices in most areas surveyed.

Based on the analysis, most of the questions show no significant differences based on gender or age, suggesting that knowledge and attitudes toward the effects of catchment land use on wetland water quality are relatively consistent across these demographics. However, on gender, Q13 and Q20 show significant differences at the 5% level, and Q3 and Q11 at the 10% level. The essential results in Q3 and Q4 for education level suggest that while there is some awareness of specific issues among certain education levels, the lack of widespread significant differences across most knowledge and attitude questions implies that many farmers might still lack comprehensive knowledge on the topic. Age(Q5) shows a substantial difference at the 5% level. Therefore, we reject the hypothesis that the lack of widespread significant differences does not support the hypothesis(H1), suggesting that farmers in Murang'a County generally do not have sufficient knowledge of the effects of catchment land use on wetland water quality.

4.5 Reliability Analysis

Reliability analysis is a critical component in the validation of a survey instrument. It ensures that the tool consistently measures the intended construct, yielding stable and reliable results. In this analysis, we focus on the internal consistency of a survey instrument composed of 20 items, evaluated using Cronbach's alpha. Cronbach's alpha measures internal consistency, reflecting the average correlation among items within a survey or questionnaire. It is expressed as a value between 0 and 1, where higher values indicate greater reliability. A commonly accepted threshold for acceptable reliability is a Cronbach's alpha of 0.70 or higher. Values below this threshold suggest potential issues with the survey items, indicating that they may not consistently

measure the same underlying construct. The reliability statistics for the survey Instrument are shown in Table 4.2;

Table 4.2: Reliability statistics

Reliability Statistics		
Cronbach's Alpha	Cronbach's Alpha Based on Standardized Items	N of Items
0.197	0.212	20

The Cronbach's alpha for the 20-item survey instrument is 0.197, significantly below the commonly accepted threshold of 0.70. This low value indicates poor internal consistency among the survey items. Several implications and potential issues arise from this finding: (i) heterogeneity of items, the items in the survey may not be measuring the same underlying construct. This lack of coherence can result from poorly worded items, items that assess different aspects of a broader concept, or the inclusion of irrelevant items. (ii) item quality, where some items may be ambiguous, confusing, or irrelevant to the measured construct. These problematic items can introduce noise into the data, lowering the overall reliability of the instrument.

The summary item statistics disclose several critical intuitions, including low mean inter-item correlation (Table 4.3). The low average correlation among items (0.013) suggests poor internal consistency, indicating that the items may not measure the same construct effectively. High Variability: The wide range and high variance in inter-item correlations indicate considerable inconsistency in how items relate. Some items are moderately positively correlated, while others show little or no negative correlation. Negative Minimum Correlation: Negative correlations (-0.393) between some items suggest that some items might be inversely related, which is problematic if all items are supposed to measure the same positive construct. Moderate Maximum Correlation: While some items show a mild positive correlation (0.683), these are not common, and this inconsistency reduces the overall reliability of the scale.

Table 4.3: Summary statistics

Summary Item Statistics							
	Mean	Minimum	Maximum	Range	Maximum / Minimum	Variance	N of Items
Inter-Item Correlations	0.013	-0.393	0.683	1.077	-1.736	0.037	20

Table 4.4 presents the item-total statistics for a 20-item survey instrument, providing insights into each item's contribution to the overall reliability.

Table 4.4: Item Total Statistics

Item-Total Statistics					
	Scale Mean if Item Deleted	Scale Variance if Item Deleted	Corrected Item-Total Correlation	Squared Multiple Correlation	Cronbach's Alpha if Item Deleted
1: I can identify wetlands in my neighborhood	35.23	6.875	0.110	0.000	0.174
2: Wetlands are important to me directly	35.17	7.316	-0.104	0.000	0.220
3: Wetlands are important to all of us	35.13	7.637	-0.331	0.000	0.251
4: Has the size of the wetland around decreased?	34.73	5.306	0.348	0.000	0.013
5: Wetlands drain into the River Mathioya	34.37	6.723	0.082	0.000	0.176
6: Cutting trees have negatively impacted on wetlands	35.03	6.240	0.014	0.000	0.217
7: The amount of agrochemicals used can be harmful to wetlands	35.40	7.421	-0.195	0.000	0.320
8. The type of agrochemicals used can be harmful to wetlands	35.27	7.168	-0.044	0.000	0.213
9. The method of applying agrochemicals can be harmful to wetlands	35.27	7.030	-0.132	0.000	0.300

10. Farmers are aware of safe agricultural practices	35.77	6.737	0.117	0.000	0.167
11. Farmers hardly care about the existence of wetlands	35.23	7.220	-0.062	0.000	0.216
12 Farmers hardly care about the type of agro chemicals they use in their farms	35.23	6.668	0.217	0.000	0.147
13. Farmers are hardly concerned about the conservation of wetlands	35.30	6.424	0.288	0.000	0.119
14. Farmers are hardly concerned about the safety of methods they use in applying agro	35.30	6.631	0.190	0.000	0.148
15 Farmers hardly find value in attending conservation awareness forums	35.23	7.013	0.040	0.000	0.191
16. Farmers hardly apply conservation methods they are aware of	35.33	6.575	0.199	0.000	0.143
17. Farmers hardly find the need to observe safety regulations when applying chemicals in	35.33	6.713	0.138	0.000	0.162
18. Farmers hardly care about the effects of agrochemicals they use on the farm on	35.37	6.723	0.123	0.000	0.166
19. Farmers are more concerned with the yields they get from the farm than the safety of	35.30	7.183	-0.057	0.000	0.218
20. Pollution of wetlands is not a concern for farmers	35.27	6.547	0.252	0.000	0.133

Reviewing the item-total statistics provides a further understanding of potential areas for improvement. Item 4 ("Has the size of the wetland around decreased?") has the highest corrected item-total correlation (0.348) and a substantially lower scale variance if deleted (5.306). Removing this item would significantly reduce Cronbach's

alpha to 0.013, indicating its positive contribution to the scale's reliability. Items 2, 3, 7, 8, 9, and 19 have negative corrected item-total correlations, suggesting they may not be aligned with the overall construct. Removing these items would likely improve the overall reliability. Items 12, 13, 16, and 20 have moderate corrected item-total correlations, indicating a reasonable fit with the overall scale, but further refinement may be needed.

Friedman's test is a non-parametric statistical test that detects treatment differences across multiple test attempts. This test is instrumental when the assumptions of ANOVA (e.g., normality) are unmet. The results indicate that Friedman's test was used to analyze the data (Table 4.5). The grand mean of the survey data is 1.85. This represents the average score across all survey items and participants, providing a baseline measure of central tendency for the dataset. The grand mean is helpful as it gives an overall picture of the average response, which can be particularly helpful when comparing different groups or conditions within the study. It also serves as a reference point against which individual item means can be compared to identify items that deviate significantly from the overall trend.

Kendall's Coefficient of Concordance (W) is 0.199, indicating a low to moderate level of agreement among the rankings of items across different participants. Kendall's W measures the degree of association or concordance between multiple rankings. A value of 0 indicates no agreement, while a 1 indicates perfect agreement. In this case, a W value of 0.199 suggests consistency in how participants ranked the items, but the deal is not strong. This low to moderate concordance may reflect differences in individual perceptions, experiences, or interpretations of the survey items, indicating that participants did not consistently rank the items similarly.

The Friedman's Chi-Square value is 119.364, and the corresponding significance (Sig.) value is 0.000. Friedman's test is a non-parametric statistical test used to detect differences in treatments across multiple test attempts, particularly useful when the assumptions of ANOVA are not met. The significance value, or p-value, indicates the probability that the observed differences in rankings occurred by chance. Since the p-value is less than 0.05, we reject the null hypothesis that there are no differences in the rankings of the items. This result suggests significant differences in how participants

ranked the items, implying that at least one item is perceived differently than the others.

The results from Friedman's test indicate significant differences in the rankings of the items across participants, as evidenced by the p-value being less than 0.05. This finding suggests that participants do not view all items as equally important or relevant, highlighting potential areas of discrepancy that warrant further investigation. However, the level of agreement among participants, as indicated by Kendall's W, is relatively low, implying that while differences in rankings exist, the overall deal is not particularly strong (Table 4.5). This variability in responses could be due to differences in individual perceptions, experiences, or interpretations of the survey items. Therefore, while the data indicate significant differences in item rankings, the weak agreement among participants suggests that these differences are not uniformly perceived. This underscores the need for further analysis and potential refinement of the survey instrument.

Table 4.5: Friedman's Test

ANOVA with Friedman's Test						
		Sum of Squares	Df	Mean Square	Friedman's Chi-Square	Sig
Between People		10.493	29	0.362		
Within People	Between Items	42.427 ^a	19	2.233	119.364	0.000
	Residual	160.173	551	0.291		
	Total	202.600	570	0.355		
Total		213.093	599	0.356		
Grand Mean = 1.85						
a. Kendall's coefficient of concordance W = 0.199.						

These scale statistics provide additional context to the overall analysis. The mean of 37.07 represents the average sum score across all 20 items for each participant. The variance of 7.237 indicates the degree of spread or dispersion of the scores around the mean, while the standard deviation of 2.690 measures the average distance of each score from the mean (Table 4.6). Given that there are 20 items on the scale, these statistics help us understand the distribution and consistency of responses.

Table 4.6: Scale Statistic

Scale Statistics			
Mean	Variance	Std. Deviation	N of Items
37.07	7.237	2.690	20

The relatively low variance and standard deviation suggest that the responses are somewhat clustered around the mean, indicating a certain level of consistency among participants (Table 4.6). However, considering the low Cronbach's alpha and the results of Friedman's test, it appears that while participants may have similar total scores, the way they rank individual items varies considerably, highlighting the complexity and potential issues with the survey's reliability and internal consistency.

4.6 Effects of catchment land use on nutrient and heavy metals inflows in Maragua and Mathioya Wetlands, Murang'a County, Kenya

Nutrient analysis in water samples from both Maragua and Mathioya wetlands found phosphate and nitrate nutrient inflows (Table 4.7). For phosphate, the minimum concentration recorded in the wetlands of Murang'a County is approximately 0.0067 mg/L, while the maximum concentration reaches 0.9177 mg/L. The mean concentration of phosphate is 0.0945 mg/L, with a standard deviation of 0.2595 mg/L (Table 4.15). This indicates a wide range of phosphate levels, suggesting variability in the sources or processes influencing phosphate distribution in the wetland ecosystem, for instance, phosphate inflows can be from geological sources as well as agricultural fertilizer inputs. The WHO as well as US EPA, have not specified recommended residue limits for phosphate inflows in drinking water.

On the other hand, nitrate levels show a minimum concentration of 1162.34 µg/L and a maximum concentration of 24142.49 µg/L. The mean nitrate concentration is 9223.37 µg/L, with a standard deviation of 6316.00 µg/L (Table 4.14). This high variance indicates substantial differences in nitrate levels across different sampling points, possibly due to varying degrees of agricultural runoff or other anthropogenic activities affecting nitrate inflows. Detected nitrate levels were below the drinking water recommended WHO and US EPA maximum residue limits of 50 mg/L and 10 mg/L (as N) $\approx$ 50 mg/L (as NO₃⁻), respectively.

Table 4.7: Nutrient inflows in water samples from Murang'a wetlands

		Nutrient analysis in water samples			
		Phosphate(s); (mg/L)		Nitrate(s); (µg/L)	
Wetland	Sampling Level	Wet season	Dry season	Wet season	Dry season
Maragua	Upstream	0.067	0.067	106524.17	11623.40
	Midstream	0.202	0.337	179867.68	82564.88
	Downstream	0.134	0.202	134900.76	167877.86
Mathioya					
	Upstream	0.067	0.202	119430.02	51949.10
	Midstream	9.176	0.404	241424.93	90870.22
	Downstream	0.134	0.337	154992.36	179684.47

The type and intensity of land use in the catchment areas have a measurable effect on the quality and composition of water entering these wetlands. These nutrient inflows confirm that farmers still apply inorganic fertilizers, that is, phosphatic and nitrogenous fertilizers, during crop production along wetland ecosystems. Massive application of such inorganic fertilizers results in residue inflows in wetland waters due to surface runoffs. Generally, there is an increase in nutrient inflows as we move downstream of the two wetland ecosystems. At the midstream level, there is a significant increase in nutrient residue levels compared to the upstream and downstream levels; this is attributed to intensive agricultural activities carried out at the midstream levels of both wetlands, which involve the application of inorganic fertilizers during crop production. Also, there are relatively higher inflows during the dry season than the wet season owing to higher precipitation, which lowers the nutrient concentration during the wet season.

The statistical analysis reveals that phosphate levels have a high variance and a positive skewness (3.452), indicating that the data distribution has a long right tail (Table 4.8). This suggests that while most sites have low to moderate phosphate levels, some sites have exceptionally high concentrations. Additionally, the kurtosis value 11.941 indicates a sharp peak in the data distribution, meaning that phosphate levels are heavily concentrated around the mean, with few extreme values. In contrast, nitrate levels exhibit high variance but have a near-zero skewness (-0.084), indicating a more symmetric distribution of nitrate concentrations around the mean. The kurtosis value of -0.024 suggests a relatively flat distribution compared to a normal distribution, implying a more even spread of nitrate levels across different sites.

Table 4.8: Nutrient summary statistics

	N Stati stic	Rang e Statis tic	Maxi mum Statis tic	Mean Statis tic	Std Erro r	Std devia tion	Varianc e	Skew ness Statis tic	Kurt osis Stati stic
Phosp hate	12	0.911	0.007	0.094 4	0.074 9	0.259	0.067	3.452	11.9 41
Nitrat e	12	22980 .151	1162. 340	12680 .916	1823. 272	6316. 001	398918 79.250	0.084	0.02 4

The high variability and significant skewness observed in phosphate levels indicate that specific sites or periods exhibit much higher phosphate concentrations than most. This could be due to localized sources of phosphate input or specific environmental conditions that favor phosphate accumulation. Nitrate levels also show considerable variability but are symmetrically distributed, suggesting a more uniform influence of nitrate sources across the wetland. The lack of significant impact of seasons, sampling stations, and sampling levels on the relationship between phosphate and nitrate concentrations implies that these factors do not play an essential role in determining nutrient levels in the wetland. The significant positive correlation between nitrate and manganese levels suggests a potential link between agricultural runoff (which often contains high levels of nitrates) and manganese concentrations in the wetland. The absence of significant correlations between phosphate and heavy metals indicates that phosphate sources might differ from nitrate sources or are less associated with metal contaminants.

a) Partial correlation analysis

The partial correlation analysis controlling for seasonal variations shows that the correlation between phosphate and nitrate concentrations is -0.229, with a p-value of 0.499 (Table 4.9). This indicates no significant relationship exists between phosphate

and nitrate levels when accounting for the effect of seasons. Therefore, seasonal changes do not significantly influence the relationship between these two nutrients.

Table 4.9: Seasonal Correlations of Nutrient Inflows

Control Variables			Phosphate	Nitrates
Seasons	Phosphate	Correlation	1.000	-0.229
		Significance (2-tailed)	0.000	0.499
		Df	0	9
	Nitrates	Correlation	-0.229	1.000
		Significance (2-tailed)	0.499	0.000
		Df	9	0

When controlling for different sampling stations, the correlation between phosphate and nitrate concentrations is -0.134, with a p-value of 0.694 (Table 4.10). This lack of significant correlation suggests that the location of sampling stations within the wetland does not significantly affect the relationship between phosphate and nitrate levels.

Table 4.10: Sampling station Correlations of nutrient inflows

Control Variables			Phosphate	Nitrates
Sampling Station	Phosphate	Correlation	1.000	-0.134
		Significance (2-tailed)	0.000	0.694
		Df	0	9
	Nitrates	Correlation	-0.134	1.000
		Significance (2-tailed)	0.694	0.000
		Df	9	0

Controlling for sampling levels (up-stream, mid-stream, down-stream) yields a correlation of 0.185 between phosphate and nitrate concentrations, with a p-value of 0.587 (Table 4.11). This again indicates no significant relationship between these two nutrients when considering the different sampling levels within the wetland.

Table 4.11: Sampling level Correlations of nutrient inflows

Control Variables			Phosphate	Nitrates
Sampling level	Phosphate	Correlation	1.000	0.185
		Significance (2-tailed)	0.000	0.587
		Df	0	9
	Nitrates	Correlation	0.185	1.000
		Significance (2-tailed)	0.587	0.000
		Df	9	0

b) Pearson correlation analysis

The Pearson correlation analysis highlights several essential relationships between phosphate, nitrates, and heavy metals in the wetlands of Murang'a County. This analysis helps to understand how these different elements interact within the wetland ecosystem, providing insights into potential sources and processes affecting nutrient and metal concentrations.

Analysis of Phosphate and heavy metals, revealed no significant correlations between phosphate levels and any of the measured heavy metals, including Cu, iron (Fe), magnesium (Mg), manganese (Mn), and zinc (Zn) as shown in Table 4.12. The p-values for these correlations ranged from 0.138 to 0.941, indicating that any observed relationships are likely due to chance rather than a true association. This lack of significant correlation suggests that the sources or processes that contribute to phosphate levels in the wetland are not the same as those that influence the concentrations of these heavy metals. It implies that phosphate inputs, which could come from agricultural runoff, wastewater discharge, or natural processes, are relatively independent of the inputs of the heavy metals measured in this study.

Nitrates and Heavy Metals (Cu, Fe, Mg, Mn, Zn): In contrast to phosphates, a significant positive correlation was found between nitrate levels and manganese (Mn) levels, with a correlation coefficient (r) of 0.653 and a p-value of 0.021. This indicates that higher concentrations of nitrates in the wetland are associated with higher levels of manganese. This significant relationship suggests a specific interaction or a common source affecting nitrates and manganese, such as agricultural runoff, which is often rich in nitrates from fertilizers and manganese from soil amendments or natural soil leaching processes. However, the correlations between nitrate levels and other heavy metals, including copper (Cu), iron (Fe), magnesium (Mg), and zinc (Zn),

were not significant. This implies that while nitrates and manganese may share familiar sources or pathways, the same cannot be said for the other heavy metals. The non-significant correlations, with p-values above the 0.05 threshold, suggest that these heavy metals in the wetland are influenced by factors or sources different from those affecting nitrate levels.

The significant positive correlation between nitrate and manganese levels indicates that specific environmental or anthropogenic factors may link the presence of these two substances. For example, agricultural practices in the region might involve inorganic fertilizers high in nitrates, which, when leached into the wetland, also mobilize manganese from the soil. This correlation could also point to common pathways such as runoff from agricultural lands or natural leaching processes where nitrates and manganese are transported together. On the other hand, the lack of significant correlations between phosphate and the measured heavy metals suggests that different processes or sources control phosphate dynamics in the wetland. Phosphate could enter the wetland from point sources like wastewater discharge, which might not significantly impact heavy metal concentrations, or from diffuse sources that do not co-occur with the heavy metals measured in this study.

Table 4.12: Elemental and nutrient Pearson Correlation analysis

		Phosp hate	Nitrat es	Cd	Cu	Fe	Mg	Mn	Zn
Phos phate	Pearson Correlatio n	1	-0.058	0.000 ^a	0.148	-0.334	0.236	-0.235	-0.029
	Sig. (2- tailed)		0.858	0.000	0.647	0.289	0.460	0.462	0.930
	N	12	12	12	12	12	12	12	12
Nitra tes	Pearson Correlatio n	-0.058	1	0.000 ^a	0.196	-0.362	0.441	0.653 [*]	0.351
	Sig. (2- tailed)	0.858		0.000	0.541	0.247	0.151	0.021	0.264
	N	12	12	12	12	12	12	12	12
Cd	Pearson Correlatio n	0.000 ^a	0.000 ^a	0.000 ^a	0.000 ^a	0.000 ^a	0.000 ^a	0.000 ^a	0.000 ^a
	Sig. (2- tailed)	0.000	0.000		0.000	0.000	0.000	0.000	0.000
	N	12	12	12	12	12	12	12	12
Cu	Pearson Correlatio n	0.148	0.196	0.000 ^a	1	-0.454	0.000	0.071	-0.195
	Sig. (2- tailed)	0.647	0.541	0.000		0.138	1.000	0.826	0.543
	N	12	12	12	12	12	12	12	12
Fe	Pearson Correlatio n	-0.334	-0.362	0.000 ^a	-0.454	1	- 0.623 [*]	-0.047	0.411
	Sig. (2- tailed)	0.289	0.247	0.000	0.138		0.030	0.885	0.184
	N	12	12	12	12	12	12	12	12
Mg	Pearson Correlatio n	0.236	0.441	0.000 ^a	0.000	- 0.623 [*]	1	0.461	0.024
	Sig. (2- tailed)	0.460	0.151	0.000	1.000	0.030		0.132	0.941
	N	12	12	12	12	12	12	12	12
Mn	Pearson Correlatio n	-0.235	0.653 [*]	0.000 ^a	0.071	-0.047	0.461	1	0.503
	Sig. (2- tailed)	0.462	0.021	0.000	0.826	0.885	0.132		0.095

	N	12	12	12	12	12	12	12	12
Zn	Pearson Correlation	-0.029	0.351	0.000 ^a	-0.195	0.411	.024	0.503	1
	Sig. (2-tailed)	0.930	0.264	0.000	0.543	0.184	.941	0.095	
	N	12	12	12	12	12	12	12	12
*. Correlation is significant at the 0.05 level (2-tailed).									
a. Cannot be computed because at least one of the variables is constant.									

Understanding these relationships is crucial for wetland management and pollution control strategies. The significant correlation between nitrates and manganese could indicate areas where agricultural practices need to be modified to reduce runoff, thereby protecting the wetland ecosystem. The lack of correlation between phosphate and heavy metals suggests that efforts to manage phosphate levels might need to focus on sources or mechanisms different from those for heavy metals.

The study indicates that nutrient inflows, particularly nitrates, in the small wetlands of Murang'a County, Kenya, are highly variable and influenced by factors such as agricultural runoff, which appears to be linked to manganese concentrations. The land use practices affecting these catchments play a critical role in shaping these wetlands' nutrient and metal dynamics. The consistent phosphate levels across different sampling points suggest a uniform distribution, likely due to the wetland's buffering capacity. More comprehensive research with larger sample sizes and detailed source tracking is recommended to understand further and mitigate the impact of nutrient and metal inflows on these ecosystems. This would help identify specific sources of contamination and develop effective strategies to protect and sustain the wetland ecosystems in Murang'a County.

c) ANOVA by seasons, sampling stations, and sampling levels

i. ANOVA Analysis by Seasons

The mean phosphate concentration during the dry season was 0.0259 mg/L (± 0.0051 mg/L), with a standard deviation of 0.0124 mg/L, indicating low variability among the samples. The 95% confidence interval for the mean ranged from 0.0129 mg/L to 0.0389 mg/L. During the dry season, phosphate levels varied between 0.0067 mg/L and 0.0405 mg/L, reflecting relatively stable and low concentrations. In contrast, phosphate concentrations during the wet season were significantly higher, with a mean

of 0.1631 mg/L ($\pm$ 0.1509 mg/L). This increase is likely driven by elevated surface runoff during heavy rainfall, which transports higher nutrient loads into the wetland. However, the wet season also showed substantial variability, evidenced by a 0.3697 mg/L standard deviation. The 95% confidence interval for the mean phosphate concentration ranged widely from -0.2249 mg/L to 0.5511 mg/L, highlighting considerable uncertainty in the mean estimate due to the broad distribution of values. Phosphate concentrations in the wet season ranged from 0.0067 mg/L to a peak of 0.9177 mg/L. When combining data from both seasons, the overall mean phosphate concentration was 0.0945 mg/L ($\pm$ 0.0749 mg/L), with a standard deviation of 0.2595 mg/L. The 95% confidence interval for the combined mean ranged from -0.0704 mg/L to 0.2593 mg/L. The full range of phosphate concentrations across all samples spanned from 0.0067 mg/L to 0.9177 mg/L, demonstrating the strong seasonal influence on phosphate variability (Figure 4.12).

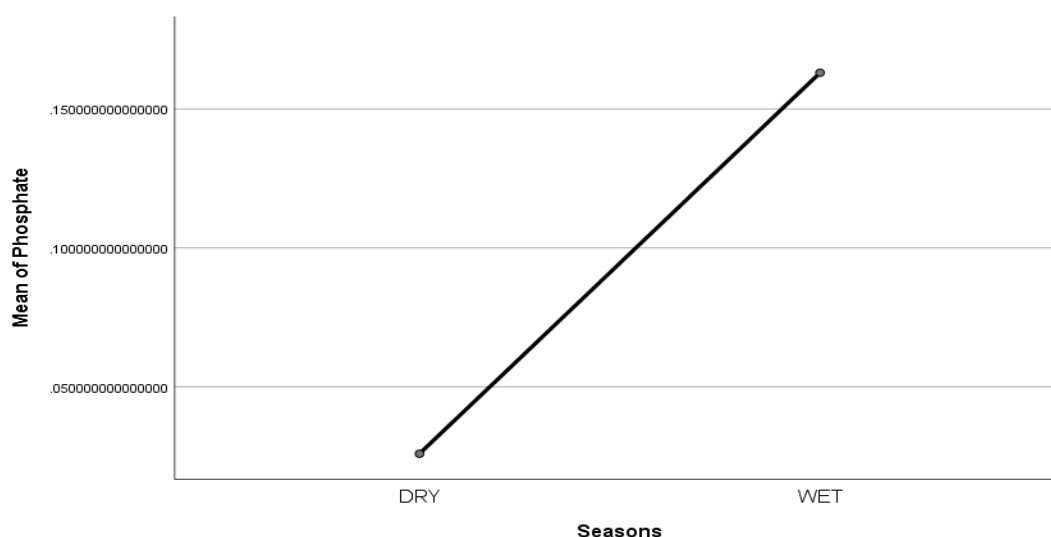


Figure 4.12: Mean seasonal concentration of Phosphate

The analysis of nitrate concentrations in the wetland ecosystem revealed considerable variability between the dry and wet seasons, with important implications for water quality and nutrient dynamics. During the dry season, the mean nitrate concentration was 9,223.37 μ g/L, with a relatively large standard deviation of 6,545.84 μ g/L. This high sample variability indicates that nitrate levels fluctuated widely during this period. Concentrations ranged from a minimum of 1,162.34 μ g/L to a maximum of 17,968.45 μ g/L, reflecting the uneven distribution of nitrates across the sampled areas. The wide 95% confidence interval for the mean, from 2,873.40 μ g/L to 15,573.34 μ g/L, further highlights the uncertainty and variability inherent in the measurements.

Such variability during the dry season could be attributed to localized nitrate inputs from sources such as groundwater seepage, point-source pollution, or residual nitrates persisting from previous seasons, especially given the reduced dilution effect due to lower water volumes.

In the wet season, the mean nitrate concentration remained consistent at 9,223.37 $\mu\text{g/L}$, but there was a notable decrease in variability, with the standard deviation falling to 4,919.00 $\mu\text{g/L}$. This suggests that nitrate concentrations became more uniform across sampling points during periods of higher water flow, likely due to increased mixing and dilution. Interestingly, the nitrate concentration range during the wet season shifted towards higher minimum values, from 10,652.42 $\mu\text{g/L}$ to 24,142.49 $\mu\text{g/L}$, indicating a general rise in nitrate levels throughout the wetland. This increase can be attributed to surface runoff that carries fertilizers, animal waste, and other nitrogen-rich substances from agricultural fields and urban areas into the water body during rainfall events. The narrower 95% confidence interval of 4,631.11 $\mu\text{g/L}$ to 13,815.64 $\mu\text{g/L}$ supports the observation of more consistent nitrate distribution during the wet season.

When data from both seasons are combined, the mean nitrate concentration remains steady at 9,223.37 $\mu\text{g/L}$, with an overall standard deviation of 6,316.00 $\mu\text{g/L}$ reflecting the combined seasonal variability. The 95% confidence interval ranges from 5,735.25 $\mu\text{g/L}$ to 12,711.50 $\mu\text{g/L}$, indicating moderate uncertainty around the mean nitrate concentration throughout the year. The total observed nitrate concentration range spans from 1,162.34 $\mu\text{g/L}$ to 24,142.49 $\mu\text{g/L}$, demonstrating significant seasonal fluctuations likely influenced by hydrological changes and varying pollutant inputs.

These findings suggest that nutrient loading, particularly during the wet season, plays a significant role in the wetland's nitrate dynamics. The elevated nitrate concentrations observed during wetter periods can largely be attributed to increased runoff, which transports nitrogenous compounds into the wetland system. While this nutrient influx supports ecosystem productivity, it raises concerns about potential eutrophication, which could harm aquatic life by triggering excessive algal growth and oxygen depletion. The decreased variability in nitrate levels during the wet season suggests that the wetland experiences more effective mixing and dilution at this time. In

contrast, during the dry season, localized sources of pollution or limited water flow may cause nitrate concentrations to vary more widely (Figure 4.13).

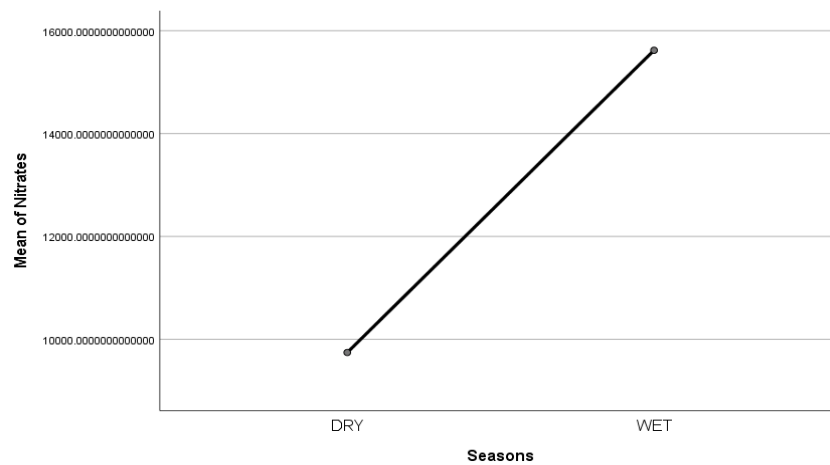


Figure 4.13: Mean seasonal concentration of Nitrate

The ANOVA analysis by seasons for phosphates has the following statistics (Table 4.13): sum of squares (SS) between groups is 0.056, SS within groups is 0.684, with a total of 0.741. the F-value is 0.825 with a significance of 0.385, which indicates insignificance. There is no significant difference in phosphate levels across different seasons, suggesting that seasonal variations do not significantly influence wetland phosphate concentrations. For nitrates, the sum of squares between groups is 103587989.131, within groups is 335222682.612, and the total is 438810671.743. Its F-value is 3.090, with a significance of 0.109. Also, there is no significant difference in nitrate levels across different seasons. However, the p-value suggests a trend that could be further explored, possibly indicating that seasonal variations might become important with a larger sample size.

Table 4.13: ANOVA Analysis of nitrates and phosphates by Seasons

		Sum of Squares	Df	Mean Square	F	Sig.
Phosphate	Between Groups	0.056	1	0.056	0.825	0.385
	Within Groups	0.684	10	0.068		
	Total	0.741	11			
Nitrates	Between Groups	103587989.13 1	1	103587989.13 1	3.090	0.109
	Within Groups	335222682.61 2	10	33522268.261		
	Total	438810671.74 3	11			

The lack of significant seasonal variation in phosphate and nitrate levels suggests that the inflows of these nutrients are relatively stable throughout the year. This could be due to consistent land use practices or the adequate buffering capacity of the wetlands against seasonal changes.

ii. ANOVA Analysis by Sampling Stations

In the Maragua wetland, the mean phosphate concentration was 0.0169 mg/L ($\pm$ 0.0042 mg/L). The standard deviation was 0.0102 mg/L, indicating low variability among the samples. The 95% confidence interval for the mean ranged from 0.0061 mg/L to 0.0276 mg/L. Phosphate concentrations in the Maragua wetland ranged from 0.0067 mg/L to 0.0337 mg/L. In the Mathioya wetland, the mean phosphate concentration was significantly higher at 0.1721 ± 0.1492 mg/L). However, there was a substantial increase in variability, with a standard deviation of 0.3655 mg/L. The 95% confidence interval for the mean ranged from -0.2115 mg/L to 0.5556 mg/L, indicating high uncertainty in the mean estimation due to the wide range of values. Phosphate concentrations in the Mathioya wetland ranged from 0.0067 mg/L to 0.9177 mg/L (Figure 4.14).

When considering both wetlands combined, the mean phosphate concentration was (0.0945 ± 0.0749 mg/L), with a standard deviation of 0.2595 mg/L. The 95% confidence interval for the mean ranged from -0.0704 mg/L to 0.2593 mg/L. The overall range of phosphate concentrations was from 0.0067 mg/L to 0.9177 mg/L.

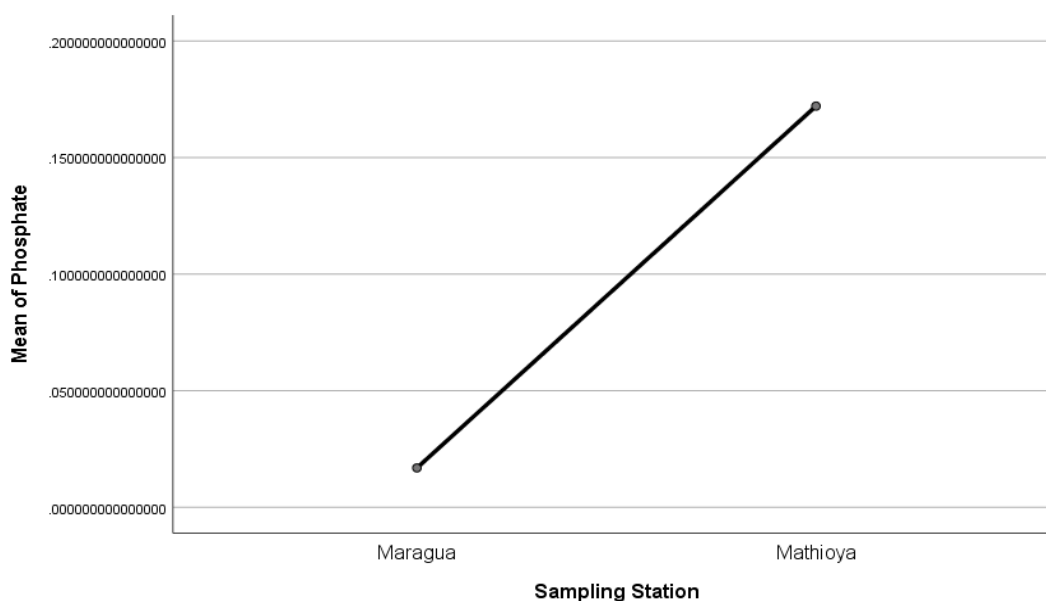


Figure 4.14: Overall mean concentrations of phosphate in sampling stations

In the Maragua wetland, nitrate concentrations varied considerably among the sampled sites. The mean nitrate concentration was 9,223.37 $\mu\text{g/L}$ with a standard error of $\pm 2,530.03 \mu\text{g/L}$, indicating substantial variability in the dataset. This variability is further reflected in the relatively high standard deviation of 6,197.28 $\mu\text{g/L}$. The 95% confidence interval for the mean nitrate concentration ranged from 4,885.66 $\mu\text{g/L}$ to 13,561.08 $\mu\text{g/L}$, demonstrating that the actual population mean is likely to fall within this interval with 95% confidence. Individual nitrate concentrations recorded in Maragua spanned a wide range, from a minimum of 1,162.34 $\mu\text{g/L}$ to a maximum of 17,986.77 $\mu\text{g/L}$, underscoring the heterogeneous distribution of nitrates within this wetland ecosystem.

Similarly, in the Mathioka River wetland, the mean nitrate concentration remained the same at 9,223.37 $\mu\text{g/L}$, but there was a slight increase in the standard error to $\pm 2,749.30 \mu\text{g/L}$. The standard deviation was also higher at 6,734.39 $\mu\text{g/L}$, indicating even greater variability in nitrate levels than in Maragua. The 95% confidence interval for Mathioka ranged from 6,905.21 $\mu\text{g/L}$ to 11,541.53 mg/L , a somewhat narrower range compared to Maragua's, suggesting slightly more consistency in the estimated mean despite the increased variation in sample values. Nitrate concentrations in Mathioka varied widely, ranging from 5,194.91 $\mu\text{g/L}$ at the lower end to 24,142.49 $\mu\text{g/L}$ at the upper extreme, highlighting areas of potentially elevated nitrate inflow or accumulation (Figur 4.15).

When data from both wetlands were combined to provide an overall picture, the mean nitrate concentration remained at 9,223.37 $\mu\text{g/L}$ with a minor standard error of $\pm 1,823.27 \mu\text{g/L}$, reflecting the increased sample size and averaging effect. The combined standard deviation was 6,316.00 $\mu\text{g/L}$, indicating significant variability across the entire dataset. The 95% confidence interval for the combined mean nitrate concentration ranged from 5,735.25 $\mu\text{g/L}$ to 12,711.50 $\mu\text{g/L}$. The full range of nitrate concentrations recorded across both wetlands extended from the lowest value of 1,162.34 $\mu\text{g/L}$ to the highest of 24,142.49 $\mu\text{g/L}$. This wide range suggests that nitrate levels can vary dramatically depending on local environmental conditions, sources of nutrient inflow, and possibly temporal factors such as seasonal runoff or anthropogenic activities.

The data indicate that nitrate contamination is present at high levels in the Maragua and Mathioya wetlands, with significant spatial variability within and between these ecosystems. This variability highlights the need for targeted management strategies that consider localized sources of nitrate pollution and the dynamic nature of nutrient distribution within wetland environments.

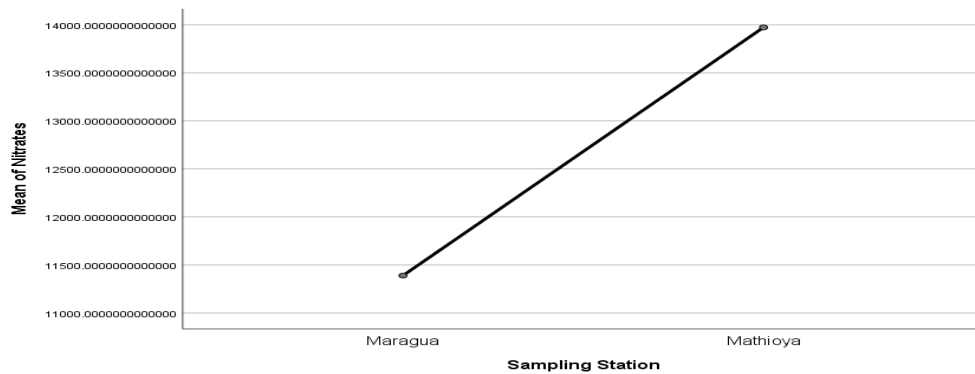


Figure 4.15: Overall mean concentrations of Nitrates in sampling stations

The Analysis of Variance (ANOVA) conducted to evaluate phosphate concentrations across different sampling stations revealed essential insights into the spatial distribution of this nutrient within the wetland ecosystem (Table 4.14). The sum of squares between groups, which measures the variability in phosphate levels attributed to differences between the sampling stations, was calculated at 0.072. In contrast, the sum of squares within groups, representing variability within each sampling station, was notably higher at 0.668. Together, these contributed to a total sum of squares of

0.741, reflecting the overall variability in phosphate concentrations observed in the collected samples. The calculated F-value from this analysis was 1.081, with an associated significance level (p-value) of 0.323. Given that this p-value exceeds the conventional threshold of 0.05, the results indicate no statistically significant difference in phosphate concentrations between the various sampling stations. The null hypothesis, which assumes that phosphate levels are the same across all stations, cannot be rejected based on the data.

These findings suggest that phosphate is distributed relatively uniformly throughout the wetland. The lack of significant spatial variation implies that the sources and inflows of phosphate into the wetland are either consistently spread across the area or that hydrological processes and mixing within the wetland help maintain a homogeneous concentration. This uniformity is essential for ecological assessment, as it points to an even nutrient load across the wetland rather than localized hotspots of phosphate accumulation. Understanding the spatial distribution of phosphate is crucial for wetland management and conservation strategies. Since phosphate levels do not significantly differ across sampling stations, interventions to manage nutrient pollution should consider the wetland as an integrated system rather than targeting specific zones. This approach can help maintain the ecological balance and prevent nutrient enrichment that could lead to problems such as eutrophication.

Table 4.14: ANOVA Analysis of nitrates and phosphates by sampling stations

		Sum of Squares	Df	Mean Square	F	Sig.
Phosphate	Between Groups	0.072	1	0.072	1.081	0.323
	Within Groups	0.668	10	0.067		
	Total	0.741	11			
Nitrates	Between Groups	20018861.372	1	20018861.372	0.478	0.505
	Within Groups	418791810.371	10	41879181.037		
	Total	438810671.743	11			

The analysis of nitrate concentrations across the different sampling stations was conducted to determine whether there were significant spatial variations within the

wetland ecosystem. The sum of squares between the groups (sampling stations) was calculated as 20,018,861.372, while the sum of squares within the groups was substantially larger at 418,791,810.371. This yielded a total sum of squares of 438,810,671.743, representing the overall variability observed in nitrate concentrations across all samples and locations. The computed F-value for this analysis was 0.478, with a corresponding significance (p-value) of 0.505. Since this p-value is greater than the commonly accepted alpha level of 0.05, the results indicate no statistically significant differences in nitrate levels between the various sampling stations. In other words, the data do not support rejecting the null hypothesis, which posits that nitrate concentrations are equal across all stations.

This lack of significant spatial variation suggests a uniform distribution of nitrate concentrations throughout the wetland. Such uniformity implies that the inflows or sources of nitrate pollution are consistent across the wetland landscape, and no particular station experiences distinctly higher or lower nitrate levels than others. Additionally, it may reflect effective mixing processes within the wetland waters, which help to homogenize nutrient concentrations despite potential localized inputs. The absence of significant differences in nitrate concentrations by sampling location indicates that factors influencing nitrate levels—such as land use practices, point source pollution, or natural nutrient cycling—are either evenly distributed or mitigated by the wetland's hydrological and ecological processes. This finding is important for understanding nutrient dynamics in the ecosystem. It suggests that management or remediation efforts targeting nitrate pollution should consider the wetland rather than focusing on specific localized areas.

iii. ANOVA Analysis by Sampling Levels

The mean phosphate concentration was measured at 0.0169 mg/L at the downstream sampling level, with a standard error of ± 0.0080 mg/L. The variability within this group was moderate, as indicated by a standard deviation of 0.0161 mg/L. The 95% confidence interval for the mean phosphate concentration at this level extended from -0.0087 mg/L to 0.0424 mg/L, suggesting some uncertainty but generally low phosphate presence. Individual sample concentrations within the downstream section ranged from a minimum of 0.0067 mg/L to a maximum of 0.0405 mg/L, reflecting a relatively narrow spread of values consistent with low phosphate levels.

Moving upstream to the midstream section, the mean phosphate concentration increased slightly to 0.0236 mg/L, with a minor standard error of ± 0.0065 mg/L, indicating more precision in the estimate than downstream. The standard deviation here was 0.0129 mg/L, lower than that found downstream, which suggests reduced variability among midstream samples. The 95% confidence interval for the mean phosphate concentration at midstream ranged between 0.0031 mg/L and 0.0442 mg/L. The observed phosphate concentrations in this section varied from 0.0067 mg/L to 0.0337 mg/L, which, although slightly higher on average than downstream, remained within a narrow and relatively low concentration range.

Significant changes were observed in the upstream section, where the mean phosphate concentration was markedly higher at 0.2429 mg/L, but this came with a substantial standard error of ± 0.2249 mg/L. Correspondingly, the standard deviation in the upstream section was 0.4499 mg/L, indicating a wide spread of phosphate concentrations across samples collected here. The 95% confidence interval for the mean phosphate concentration upstream was broad, ranging from -0.4729 mg/L to 0.9587 mg/L, reflecting high uncertainty due to the significant variability. Individual phosphate concentrations in this section showed a substantial range, from a minimum of 0.0135 mg/L to a maximum of 0.9177 mg/L, suggesting that specific locations upstream may experience elevated phosphate inputs or conditions favoring phosphate accumulation (Figure 4.16).

When considering all sampling sections combined, the overall mean phosphate concentration was 0.0945 mg/L with a standard error of ± 0.0749 mg/L. The combined data set showed a relatively high standard deviation of 0.2595 mg/L, indicative of the combined variability across all sampling points. The 95% confidence interval for the overall mean ranged from -0.0704 mg/L to 0.2593 mg/L. Across all samples, phosphate concentrations spanned a wide range from a low of 0.0067 mg/L to a high of 0.9177 mg/L, underscoring considerable heterogeneity in phosphate levels throughout the studied wetland system.

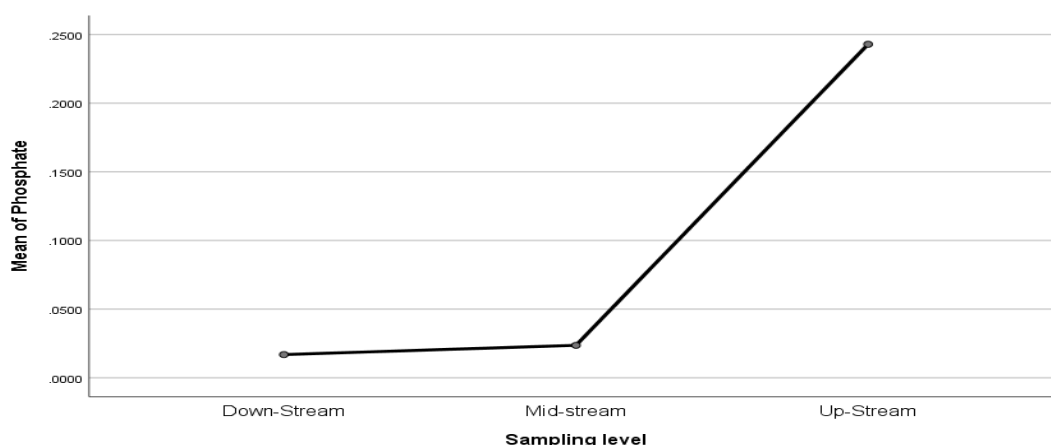


Figure 4.16: Mean concentration of Phosphate at sampling levels

The mean nitrate concentration Downstream was 9223.37 $\mu\text{g/L}$ ($\pm 591.20 \mu\text{g/L}$). The standard deviation was 1182.41 $\mu\text{g/L}$, indicating moderate variability among the samples. The 95% confidence interval for the mean could not be calculated adequately as all samples shared the same mean. Nitrate concentrations in the downstream section ranged from 15499.24 $\mu\text{g/L}$ to 17986.77 $\mu\text{g/L}$. The mean nitrate concentration for the Mid-Stream was 9223.37 $\mu\text{g/L}$ ($\pm 4792.33 \mu\text{g/L}$), with a standard deviation of 9223.37 $\mu\text{g/L}$. The 95% confidence interval for the mean ranged from -3280.85 $\mu\text{g/L}$ to 21727.60 $\mu\text{g/L}$. Nitrate concentrations in the mid-stream section ranged from 1162.34 $\mu\text{g/L}$ to 24142.49 $\mu\text{g/L}$. The mean nitrate concentration Up-Stream was 9011.70 $\mu\text{g/L}$ ($\pm 1483.88 \mu\text{g/L}$). The standard deviation was 2967.76 $\mu\text{g/L}$. The 95% confidence interval for the mean ranged from 4289.34 $\mu\text{g/L}$ to 13734.06 $\mu\text{g/L}$. Nitrate concentrations in the upstream section ranged from 5194.91 $\mu\text{g/L}$ to 11943.00 $\mu\text{g/L}$. Considering all sections combined, the mean nitrate concentration was 9223.37 $\mu\text{g/L}$ ($\pm 1823.27 \mu\text{g/L}$), with a standard deviation of 6316.00 $\mu\text{g/L}$ (Figure 4.10). The 95% confidence interval for the mean ranged from 5735.25 $\mu\text{g/L}$ to 12711.50 $\mu\text{g/L}$. The overall range of nitrate concentrations was from 1162.34 mg/L to 24142.49 $\mu\text{g/L}$.

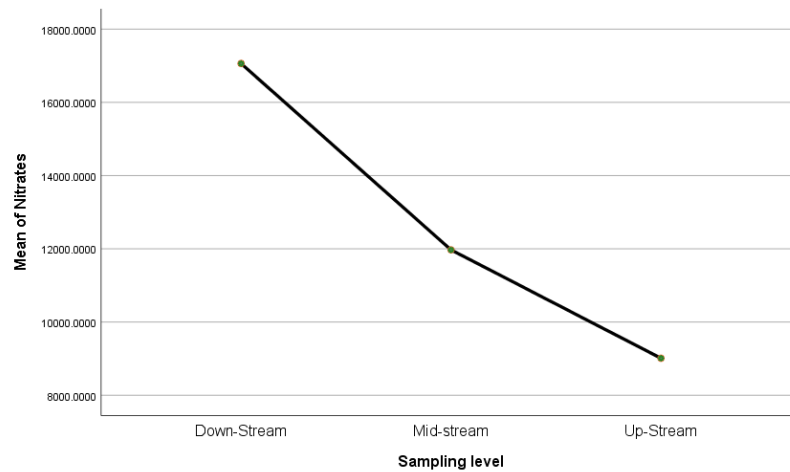


Figure 4.17: Mean concentration of Nitrate at sampling levels

The results indicate that phosphate concentrations vary significantly between different river sections, with the highest concentrations observed upstream and the lowest downstream. Nitrate concentrations showed no clear trend between the sections but exhibited substantial variability. The data highlight the impact of geographical and possibly anthropogenic factors on nutrient levels in these river sections, which could have implications for water quality management and ecological health assessments. The ANOVA analysis for phosphate and nitrate concentrations across different sampling levels (up-stream, mid-stream, down-stream) revealed non-significant differences between sampling levels, suggesting that nutrient levels are consistent throughout the flow of the wetlands.

Post Hoc Multiple Comparisons (LSD)

In statistical analysis, Least Significant Difference (LSD) is a post-hoc test used after an Analysis of Variance (ANOVA) to determine whether the difference between two specific group means is statistically significant. After ANOVA indicates that at least one group mean is different, LSD helps pinpoint which pairs of means are significantly different from each other. The post hoc multiple comparisons using the Least Significant Difference (LSD) method for phosphate levels demonstrated no significant differences among the downstream, mid-stream, and upstream sampling points (Table 4.15). Specifically, the mean difference between down-stream and mid-stream was -0.0067 with a p-value of 0.972, indicating no significant variation. Similarly, the mean difference between down-stream and up-stream was -0.2260 with a p-value of 0.250, and between mid-stream and up-stream, the mean difference was -0.2193 with a p-value of 0.263. These results indicate that phosphate concentrations are uniformly

distributed across different parts of the wetland, supporting the conclusion that there is no significant spatial variation in phosphate levels.

Table 4.15: LSD Comparisons for Nutrients

Multiple Comparisons							
LSD							
Dependent Variable	(I) Sampling level	(J) Sampling level	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
						Lower Bound	Upper Bound
Phosphate	Down-Stream	Mid-stream	-0.00674	0.1838	0.972	-0.4226	0.4091
		Up-Stream	-0.2260	0.1838	0.250	-0.6419	0.1898
	Mid-stream	Down-Stream	0.0067	0.1838	0.972	-0.40913	0.4226
		Up-Stream	-0.2192	0.1838	0.263	-0.6351	0.1966
	Up-Stream	Down-Stream	0.22604	0.1838	0.250	-0.1898	0.6419
		Mid-stream	0.2192	0.1838	0.263	-0.1966	0.6352
Nitrates	Down-Stream	Mid-stream	5090.0763	4124.5488	0.248	-4240.3013	9223.3720
		Up-Stream	8048.8550	4124.5488	0.083	-1281.5226	9223.3720
	Mid-stream	Down-Stream	-5090.0763	4124.5488	0.248	-9223.3720	4240.3013
		Up-Stream	2958.7786	4124.5488	0.491	-6371.5990	9223.3720
	Up-Stream	Down-Stream	-8048.8550	4124.5488	0.083	-9223.3720	1281.5226
		Mid-stream	-2958.7786	4124.5488	0.491	-9223.3720	6371.5990

The post hoc comparisons for nitrate levels also revealed no significant differences between the sampling points. However, there was a noticeable trend towards significance in the comparison between downstream and upstream sampling points. The mean difference between the down-stream and mid-stream was 5090.076 with a p-value of 0.248, indicating no significant difference. The mean difference between down-stream and up-stream was 8048.855 with a p-value of 0.083, suggesting a potential trend towards significance. Lastly, the mean difference between mid-stream and up-stream was 2958.779, with a p-value of 0.491. The p-value of 0.083 for the downstream against upstream comparison suggests that this difference might become significant with a larger sample size, indicating potential spatial variation in nitrate levels within the wetland.

In conclusion, the ANOVA analyses consistently indicate no statistically significant differences in the levels of phosphate and nitrates based on seasons, sampling stations, or sampling levels. The p-values for all comparisons exceed the significance threshold of 0.05, suggesting that these factors do not strongly influence the variations in nutrient levels. However, the trends observed, particularly in nitrate levels between different sampling levels and stations, imply that further investigation with a larger sample size might reveal significant differences. These findings suggest that nutrient inflows into the wetlands are relatively stable and uniform across different spatial and temporal scales, potentially due to consistent land use practices and the adequate buffering capacity of the wetlands. This stability is crucial for maintaining the ecological balance of the wetlands and ensuring that nutrient levels do not reach harmful concentrations. However, the potential trends observed warrant further investigation to understand better the spatial dynamics of nutrient distribution and their potential impact on wetland health and function.

iv. The hypothesis evaluations

Hypothesis 2 stated differences in the level of nutrients, heavy metals, and pesticide inflows on small wetland ecosystems in Murang'a County. Based on the statistical analyses performed, including ANOVA and LSD, we must determine whether to reject or fail to reject the hypothesis H_{02} . The study of phosphate levels using ANOVA revealed no significant differences between different sampling stations or sampling levels, namely upstream, mid-stream, and downstream. This result indicates that

phosphate concentrations are relatively uniform across the various parts of the wetland ecosystem. Further supporting this finding, the post hoc comparisons confirmed no significant differences in phosphate levels among the different sampling points. These consistent results suggest that the inflow of phosphates into the wetland is evenly distributed, regardless of the specific location within the wetland.

Similarly, the ANOVA results for nitrate levels showed no significant differences between sampling stations or sampling levels. This indicates a uniform distribution of nitrates throughout the wetland. However, the post hoc comparisons did reveal a potential trend towards significance between the downstream and upstream levels. The observed trend suggests that with a larger sample size, there might be detectable spatial variations in nitrate levels. Despite this trend, the current data do not provide strong enough evidence to confirm significant differences in nitrate concentrations across the sampling points.

The hypothesis H_{02} posits that there are differences in the levels of nutrients (phosphates and nitrates), heavy metals, and pesticide inflows in small wetland ecosystems in Murang'a County. However, the statistical analyses performed for nutrients, phosphates explicitly, and nitrates indicate no significant differences in their levels across different sampling stations and levels. The p-values for phosphates and nitrates were above the significance threshold 0.05, suggesting that any observed differences are not statistically significant. Furthermore, the post hoc LSD tests reinforced these findings by showing no significant mean differences between the sampling points.

Based on the provided data and analyses, we accept the hypothesis H_{02} about nutrients (phosphates and nitrates). This means that the data do not provide sufficient evidence to support the claim that there are differences in the levels of these nutrients across different parts of the wetland ecosystems in Murang'a County. The findings suggest that the nutrient inflows into the wetlands are relatively stable and uniform across various spatial and temporal scales. This uniformity could be attributed to consistent land use practices and the adequate buffering capacity of the wetlands. Nonetheless, the potential trend observed in nitrate levels warrants further investigation with a larger sample size to determine any spatial variations conclusively.

4.7 Elemental inflows in water samples

Elemental analysis of water samples from both Maragua and Mathioya wetlands identified copper (Cu), iron (Fe), magnesium (Mg), manganese (Mn), zinc (Zn) and lead (Pb) to be present at varied concentrations downstream during both the wet and dry seasons as shown in Table 4.23. Copper concentrations ranged from 0.000 mg/L to 0.008 mg/L, with a mean concentration of 0.0028 mg/L. The slight positive skewness (0.487) suggests that the distribution of copper concentrations is slightly skewed to the right, indicating that a few samples exhibited higher copper levels than the majority. The low kurtosis value (0.005) indicates a distribution relatively close to a normal distribution.

Iron concentrations displayed a wider range, from 0.000 mg/L to 0.063 mg/L, with an average concentration of 0.0246 mg/L. The positive skewness (0.446) suggests a slight skew to the right in the distribution, indicating that some samples had higher iron concentrations. However, the negative kurtosis (-1.219) indicates a relatively flat distribution compared to a normal distribution. Magnesium concentrations exhibited a narrow range, from 0.00003 mg/L to 0.0041 mg/L, with a mean concentration of 0.0019 mg/L. The skewness value close to zero (-0.045) suggests a distribution close to symmetry, indicating a relatively balanced spread of magnesium concentrations across the samples. Moreover, the negative kurtosis value (-1.834) indicates a flatter distribution compared to a normal distribution.

Manganese concentrations varied significantly, ranging from 0.000 mg/L to 0.029 mg/L, with a mean concentration of 0.0055 mg/L. The highly positive skewness (2.953) and kurtosis (9.519) values indicate a distribution with a long right tail and a sharp peak, respectively. This suggests that while most samples exhibited low manganese concentrations, a few had notably higher levels. Zinc concentrations ranged from 0.0043 mg/L to 0.041 mg/L, with a mean concentration of 0.0198 mg/L. The slight positive skewness (0.451) indicates a mild right skewness in the distribution, implying that some samples had slightly higher zinc concentrations. The negative kurtosis value (-0.536) suggests a flatter distribution compared to a normal distribution.

Cadmium, which is one of the hazardous heavy metals, was also analyzed to confirm the possibility of its existence. However, Cadmium concentrations in the wetland

samples were consistently recorded at zero across all measurements, as shown in Table 4.16. This constant value indicates either the absence of cadmium or that its concentration falls below the detection limit of the measurement method. Consequently, no variability in cadmium levels was observed within the sampled area. Lead, which is also, one of the most hazardous heavy metals, was found to have a mean concentration of 0.2625 mg/L and a normal distribution downstream.

Table 4.16: Seasonal elemental analysis of water samples from both Maragua and Mathioya wetland ecosystems

		Elemental analysis of water													
		Concentration (mg/L)													
Wetland	Sampling Level	Cd		Cu		Fe		Mg		Mn		Zn		Pb	
		Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry
Maragua	Upstream	0	0	0.02044	0	0.183384	0	0.008984	0.000281	0.009627	0	0.043132	0.060714	0.218112	0.081633
	Midstream	0	0	0.218112	0.041405	0.236069	0.438703	0.016189	0.024799	0.042118	0.02136	0.405495	0.142857	0.480867	0.118622
	Downstream	0	0	0.080189	0.039832	0.227964	0.583587	0.000281	0.000561	0.036402	0.040012	0.175549	0.289286	0.454082	0.096939
Mathioya	Upstream	0	0	0.041405	0	0.62614	0	0.030039	0.000281	0	0.029182	0.191209	0.148077	0.039541	0.142857
	Midstream	0	0	0.042453	0.002096	0.187437	0.471125	0.041456	0.033034	0.292118	0.064681	0.359615	0.253846	0.577806	0.22449
	Downstream	0	0	0.045597	0	0	0	0.034157	0.032566	0.055656	0.065884	0.09011	0.221703	0.5	0.214286

The trend in residue concentrations along the wetlands is an increase in elemental inflows as we moved from upstream to downstream of the two wetland ecosystems. There was a significant increase in elemental residue levels at the midstream level compared to the upstream and downstream sampling points. This is attributed to intensive agricultural practices along the wetlands' midstream regions, contributing to inflow build-up via surface runoffs. Trace amounts of some metal ions, such as zinc (Zn), copper (Cu), iron (Fe), magnesium (Mg) and manganese (Mn), are essential for metabolic processes in living organisms. However, excessive bioaccumulation in tissues may reach toxic levels, posing health hazards to living organisms. Copper (II)

ions residues are associated with applying copper-based fungicides used by farmers for crop protection where inflows get into wetland ecosystems through surface run-offs. All elemental inflows detected were below the permissible limits for drinking water, except for lead, whose concentration, across the seasons, surpassed the permissible residue limits of 0.01 mg/L and 0.015 mg/L set by WHO¹⁸ and US EPA¹⁹, respectively. These findings are similar to results from a research carried out at Sosiani River in Eldoret where all other metals were within the permissible limits in drinking water except for lead metal whose concentration was at 0.106 mg/L which is above the WHO set limit. This raises an urgent concern in investigating the possible sources of lead in wetland waters from Murang'a County.

The descriptive statistics underscore the variability in heavy metal concentrations within the wetland ecosystem (Table 4.17). While cadmium was consistently absent or below detection limits, copper, iron, magnesium, manganese, and zinc displayed varying concentration and distribution patterns. These variations highlight the complex interplay of environmental factors influencing the presence and distribution of heavy metals in the wetland samples.

Table 4.17: Descriptive statistics for elemental Analysis

Element	N	Statistic	Range	Min	Max	Mean	Std Deviation	Variance	Skewness	Kurtosis
Cd	12		0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Cu	12		0.008	0.000	0.008	0.003	0.002	0.000	0.487	0.005
Fe	12		0.063	0.000	0.063	0.025	0.023	0.000	0.446	-1.219
Mg	12		0.041	0.000	0.044	0.002	0.002	0.000	0.045	-1.834
Mn	12		0.029	0.000	0.029	0.005	0.008	0.000	2.953	9.519
Zn	12		0.036	0.004	0.041	0.019	0.011	0.000	0.451	-0.536

4.7.1 Correlation analysis for heavy metals

i. Pearson correlation

The Pearson correlation analysis was performed to compute and present the Pearson correlation coefficients for each pair of variables and test the statistical significance of such correlations. The importance of each correlation was tested using a two-tailed test, which assesses whether the observed correlation differs significantly from zero, indicating a proper relationship between the variables. Descriptive statistics were also computed to provide additional context, although these specific statistics are not detailed in the dataset. The pairwise deletion was employed to handle missing data, ensuring each correlation calculation used all available data points without introducing bias from incomplete cases. The results of the correlations indicate that the correlation coefficients involving Cd could not be computed. This is likely because Cd was a constant variable in this dataset. A continuous variable does not vary and cannot be correlated with other variables.

The correlation analysis reveals that the relationship between Copper (Cu) and Iron (Fe) has a Pearson correlation coefficient of -0.454 with a p-value of 0.138 . This indicates a moderate negative relationship; however, it is not statistically significant, suggesting that changes in Fe concentration are not reliably associated with changes in Cu concentration in this dataset. The correlation between Cu and Mg is exactly 0.0000 , with a p-value of 1.000 , showing no relationship whatsoever. For Manganese (Mn), the correlation with Cu is 0.071 with a p-value of 0.826 , indicating a very weak positive relationship that is not statistically significant. Finally, the correlation between Cu and Zn is -0.195 with a p-value of 0.543 , which means a weak negative relationship, again not statistically significant.

Iron (Fe) shows a significant correlation with Magnesium (Mg). The correlation coefficient is -0.623 , with a p-value of 0.030 , indicating a strong negative relationship. This means that as the concentration of Fe increases, the concentration of Mg tends to decrease, and vice versa. This finding is statistically significant at the 0.05 level. On the other hand, the correlation between Fe and Manganese (Mn) is -0.047 with a p-value of 0.885 , indicating no significant relationship. The correlation between Fe and Zn is 0.4110 , with a p-value of 0.184 , showing a moderate positive relationship, but this is not statistically significant.

Mg exhibits a moderate positive correlation with Mn, with a correlation coefficient of 0.4610 and a p-value of 0.132. This relationship is close to being statistically significant, suggesting a potential trend where higher Mg concentrations might be associated with higher Mn concentrations, but this result is not definitive. The correlation between Mg and Zn is 0.024 with a p-value of 0.941, suggesting no meaningful relationship between these two variables.

The correlation between Manganese (Mn) and Zn is 0.5030, with a p-value of 0.095. This indicates a moderate positive relationship that approaches statistical significance. This suggests that there might be a trend where higher Mn concentrations are associated with higher Zn concentrations. Still, the evidence is insufficient to confirm this with high confidence.

The analysis revealed (i) significant correlation: The only statistically significant correlation found in this dataset is the negative correlation between (Fe) and (Mg) ($r^2 = -0.623$, $p = 0.030$). This strong inverse relationship suggests that an increase in Fe concentration is associated with a decrease in Mg concentration and vice versa. This finding could have important implications for studies investigating the interaction between these two metals in various environments or biological systems. (ii) non-significant correlations: The correlations between (Cu) and the other metals (Fe, Mg, Mn, Zn) were not statistically significant, indicating that within this dataset, Cu does not show a strong linear relationship with any of these metals. Similarly, the correlations involving Manganese (Mn) with Iron (Fe) and Zinc (Zn) were also not significant, suggesting that Mn does not have a strong linear relationship with these metals in this sample. The correlation between (Mg) and (Mn) ($r^2 = 0.461$, $p = 0.132$) is close to significance, suggesting a potential trend that might be worth exploring further with a larger sample size. The correlation analysis indicates that most of the variables do not exhibit significant linear relationships with each other, except for the negative correlation between Fe and Mg (Table 4.25). This considerable finding underscores a specific interaction that warrants further investigation. However, the constant nature of Cd in this dataset prevents any meaningful analysis involving this metal.

ii. Partial correlation for elemental analysis

a) Influence of seasons, sampling stations, and sampling levels

The first analysis controlled for seasons, sampling stations, and sampling levels to understand how these factors influence the correlations among the metals. This comprehensive control aimed to isolate the intrinsic relationships between the metals, removing the effects of location and depth of sampling (Table 4.25).

Cd again showed perfect correlation with itself and no correlation with other metals. This pattern, as before, might indicate that Cd levels were consistent across different conditions or that the variable's nature prevented meaningful correlation calculations. Cu had a correlation of -0.314 with Fe and a p-value of 0.410 , suggesting a moderate negative relationship that is not significant. The correlation between Cu and Mg was 0.077 , with a p-value of 0.843 , indicating a weak positive relationship. The correlations between Cu and Mn and Cu and Zn were 0.029 and -0.339 , with p-values of 0.941 and 0.372 , respectively, indicating no significant relationships.

Fe had a significant negative correlation with Mg (-0.682 , $p = 0.043$), reinforcing the strong inverse relationship observed in the previous analysis. This suggests that higher Fe concentrations are consistently associated with lower Mg concentrations regardless of season, location, and depth. The correlation between Fe and Mn was $0.1660.1660.166$ with a p-value of 0.670 , indicating no significant relationship. The correlation between Fe and Zn was 0.571 with a p-value of 0.108 , indicating a moderate positive relationship that is not statistically significant but is close to being noteworthy. Mg had a correlation of 0.226 with Mn and a p-value of 0.558 , showing no significant relationship. The correlation between Mg and Zn was -0.113 , with a p-value of 0.773 , indicating a weak negative relationship. Mn had a correlation of 0.481 with Zn and a p-value of 0.190 , suggesting a moderate positive relationship that is not statistically significant. This aligns with the previous analysis, indicating a potential but non-significant trend of co-variation between Mn and Zn concentrations under controlled conditions of seasons, sampling stations, and levels (Table 4.18).

Table 4.18: Partial correlation for elemental analysis

Control Variables			Cd	Cu	Fe	Mg	Mn	Zn
Seasons & Samplin g Station & Samplin g level	Cd	Correlation	1.000	0.000	0.000	0.000	0.000	0.000
		Significance (2-tailed)	0.000	0.000	0.000	0.000	0.000	0.000
		Df	0	7	7	7	7	7
	Cu	Correlation	0.000	1.000	-0.314	0.077	0.029	-0.339
		Significance (2-tailed)	0.000	0.000	0.410	0.843	0.941	0.372
		Df	7	0	7	7	7	7
	Fe	Correlation	0.000	-0.314	1.000	-0.682	0.166	0.571
		Significance (2-tailed)	0.000	0.410	0.000	0.043	0.670	0.108
		Df	7	7	0	7	7	7
	Mg	Correlation	0.000	0.077	-0.682	1.000	0.226	-0.113
		Significance (2-tailed)	0.000	0.843	0.043	0.000	0.558	0.773
		Df	7	7	7	0	7	7
	Mn	Correlation	0.000	0.029	0.166	0.226	1.000	0.481
		Significance (2-tailed)	0.000	0.941	0.670	0.558	0.000	0.190
		Df	7	7	7	7	0	7
	Zn	Correlation	0.000	-0.339	0.571	-0.113	0.481	1.000
		Significance (2-tailed)	0.000	0.372	0.108	0.773	0.190	0.000
		Df	7	7	7	7	7	0

b) Influence of Seasons

In the second partial correlation analysis, we controlled for the variable "Seasons" to investigate the relationships among Cd, Cu, Fe, Mg, Mn, and Zn. The results show how seasonal variations might impact the correlations between these metals. As expected, Cd showed a perfect correlation with itself, but the correlations with other metals could not be computed. This proposes that the Cd data structure did not allow for meaningful correlations to be established when seasons were controlled.

Cu showed varying degrees of correlation with other metals. The correlation between Cu and Iron (Fe) was -0.236 with a p-value of 0.485 , indicating a weak negative relationship that is not statistically significant. Similarly, the correlation between Cu and Mg was -0.16 , with a p-value of 0.624 , suggesting an even weaker negative relationship. The correlation between Cu and Mn was -0.094 with a p-value of 0.784 ,

also not significant. Lastly, the correlation between Cu and Zn was -0.332 with a p-value of 0.319 , indicating a moderate negative relationship, though still not statistically significant. Fe exhibited a significant negative correlation with Mg, with a coefficient of -0.606 and a p-value of 0.048 , indicating a strong inverse relationship when seasons were controlled. This suggests that as the concentration of Fe increases, the concentration of Mg tends to decrease, and vice versa, independent of seasonal effects. The correlation between Fe and Mn was 0.080 , with a p-value of 0.816 , showing no significant relationship. The correlation between Fe and Zn was 0.535 with a p-value of 0.090 , indicating a moderate positive relationship close to significance.

Mg had a correlation of 0.431 with Mn with a p-value of 0.185 , indicating a moderate positive relationship that is not statistically significant. The correlation between Mg and Zn was nearly zero (-0.001) with a p-value of 0.998 , showing no relationship. Mn and Zn correlated 0.4930 with a p-value of 0.123 , indicating a moderate positive relationship that is not statistically significant. This suggests that higher concentrations of Mn might be associated with higher concentrations of Zn. However, this trend is not strong enough to be considered statistically significant under the influence of seasons.

c) Influence of Sampling Stations

In the third analysis, we controlled for the variable "Sampling Stations" to assess the influence of geographic location on the relationships among the metals. This control aimed to determine if the spatial distribution of sampling points affected the observed correlations. Cadmium (Cd) once again showed no correlations with other metals, highlighting the persistent issue of Cd being a constant or uncorrelated variable in this dataset (Table 4.19).

Copper (Cu) had a correlation of -0.515 with Fe with a p-value of 0.105 , indicating a moderate negative relationship that approaches statistical significance. This suggests that spatial factors might influence the negative relationship between Cu and Fe. The correlation between Cu and Mg was 0.241 , with a p-value of 0.476 , indicating a weak positive relationship. The correlation between Cu and Mn was 0.199 with a p-value of 0.557 , and the correlation between Cu and Zn was -0.173 with a p-value of 0.612 , indicating no significant relationships. Fe showed a significant negative correlation

with Mg (-0.711 , $p = 0.014$), consistent with the previous analyses, underscoring the strong inverse relationship between these two metals across different locations. The correlation between Fe and Mn was 0.011 , with a p -value of 0.973 , indicating no relationship. The correlation between Fe and Zn was 0.435 with a p -value of 0.181 , showing a moderate positive relationship that is not statistically significant.

Table 4.19: Correlation of the influence of sampling stations on elemental analysis

Correlations								
Control Variables			Cd	Cu	Fe	Mg	Mn	Zn
Sampling Station	Cd	Correlation	1.000	0.000	0.000	0.000	0.000	0.000
		Significance (2-tailed)	0.000	0.000	0.000	0.000	0.000	0.000
		Df	0	9	9	9	9	9
	Cu	Correlation	0.000	1.000	-0.515	0.241	0.199	-0.173
		Significance (2-tailed)	0.000	0.000	0.105	0.476	0.557	0.612
		Df	9	0	9	9	9	9
	Fe	Correlation	0.000	-0.515	1.000	-0.711	0.011	0.435
		Significance (2-tailed)	0.000	0.105	0.000	0.014	0.973	0.181
		Df	9	9	0	9	9	9
	Mg	Correlation	0.000	0.241	-0.711	1.000	0.287	-0.068
		Significance (2-tailed)	0.000	0.476	0.014	0.000	0.393	0.842
		Df	9	9	9	0	9	9
	Mn	Correlation	0.000	0.199	0.011	0.287	1.000	0.503
		Significance (2-tailed)	0.000	0.557	0.973	0.393	0.000	0.115
		Df	9	9	9	9	0	9
	Zn	Correlation	0.000	-0.173	0.435	-0.068	0.503	1.000
		Significance (2-tailed)	0.000	0.612	0.181	0.842	0.115	0.000
		Df	9	9	9	9	9	0

Mg had a correlation of 0.287 with Mn with a p -value of 0.393 , indicating a weak positive relationship that is not significant. The correlation between Mg and Zn was -0.068 , with a p -value of 0.842 , indicating no relationship. Mn had a correlation of 0.503 with Zn and a p -value of 0.115 , suggesting a moderate positive relationship close to significance. This trend aligns with previous observations, indicating a consistent but non-significant co-variation between Mn and Zn across different sampling stations.

d) Influence of sampling levels

The final analysis controlled for the variable "Sampling Levels" to examine how the depth or intensity of sampling affected the correlations among the metals. This control aimed to identify whether variations in sampling methodology influenced the observed relationships. Cd continued to show no correlations with other metals, reinforcing the earlier findings of Cd being a constant or uncorrelated variable. Cu correlated -0.456 with Fe with a p-value of 0.158 , indicating a moderate negative relationship that is not statistically significant. This suggests that sampling depth might not significantly impact the relationship between Cu and Fe. The correlation between Cu and Mg was -0.002 with a p-value of 0.995 , showing no relationship. The correlations between Cu and Mn and Cu and Zn were 0.0580 and -0.212 , with p-values of 0.864 and 0.531 , respectively, indicating no significant relationships.

Fe showed a significant negative correlation with Mg (-0.624 , $p = 0.040$), consistent with previous findings, suggesting a robust inverse relationship between these two metals that persists regardless of sampling depth. The correlation between Fe and Mn was -0.048 , with a p-value of 0.889 , indicating no relationship. The correlation between Fe and Zn was 0.416 with a p-value of 0.204 , showing a moderate positive relationship that is not statistically significant. Mg had a correlation of 0.462 with Mn with a p-value of 0.152 , indicating a moderate positive relationship that is not statistically significant but suggests a trend worth further exploration. The correlation between Mg and Zn was 0.021 , with a p-value of 0.952 , indicating no relationship.

Mn had a correlation of 0.493 with Zn and a p-value of 0.123 , suggesting a moderate positive relationship that is not statistically significant. This result is consistent with previous analyses, indicating a potential trend of co-variation between Mn and Zn concentrations across different sampling levels. In summary, the partial correlation analyses across different control variables (Seasons, Sampling Stations, and Sampling Levels) revealed several consistent patterns and relationships among the metals. A key finding across all analyses was the significant negative correlation between Fe and Mg. This inverse relationship was consistently strong and statistically significant, suggesting that higher concentrations of Fe are associated with lower concentrations of Mg, regardless of seasonal, spatial, or sampling level variations. This relationship

might indicate underlying geochemical or biological processes that regulate the distribution and interaction of these metals.

Cu generally showed weak to moderate negative correlations with other metals, but none of these relationships were statistically significant across the different analyses. This suggests that Cu might not have strong direct interactions with Fe, Mg, Mn, or Zn in the contexts of the water samples examined. Mn and Zn consistently showed moderate positive correlations that approached but did not reach statistical significance (Table 4.20).

Table 4.20: Correlation on the influence of sampling levels on elemental analysis

Control Variables			Cd	Cu	Fe	Mg	Mn	Zn
Sampling level	Cd	Correlation	1.000	0.000	0.000	0.000	0.000	0.000
		Significance (2-tailed)	0.000	0.000	0.000	0.000	0.000	0.000
		Df	0	9	9	9	9	9
	Cu	Correlation	0.000	1.000	-0.456	-0.002	0.058	-0.212
		Significance (2-tailed)	0.000	0.000	0.158	0.995	0.864	0.531
		Df	9	0	9	9	9	9
	Fe	Correlation	0.000	-0.456	1.000	-0.624	-0.048	0.416
		Significance (2-tailed)	0.000	0.158	0.000	0.040	0.889	0.204
		Df	9	9	0	9	9	9
	Mg	Correlation	0.000	-0.002	-0.624	1.000	0.462	0.021
		Significance (2-tailed)	0.000	0.995	0.040	0.000	0.152	0.952
		Df	9	9	9	0	9	9
	Mn	Correlation	0.000	0.058	-0.048	0.462	1.000	0.493
		Significance (2-tailed)	0.000	0.864	0.889	0.152	0.000	0.123
		Df	9	9	9	9	0	9
	Zn	Correlation	0.000	-0.212	0.416	0.021	0.493	1.000
		Significance (2-tailed)	0.000	0.531	0.204	0.952	0.123	0.000
		Df	9	9	9	9	9	0

4.7.2 ANOVA Analysis for Heavy Metals

a) ANOVA for metal concentrations by seasons

The Analysis of Variance (ANOVA) was conducted to examine whether the concentrations of six metals, Cadmium (Cd), Copper (Cu), Iron (Fe), Magnesium (Mg), Manganese (Mn), and Zinc (Zn) differ significantly across different seasons (Table 4.21). The null hypothesis (H_0) assumed no significant differences in metal concentrations between seasons exist. In contrast, the alternative hypothesis (H_1) posited that at least one season exhibits a significantly different metal concentration than others. For Cadmium, the ANOVA could not be meaningfully performed due to insufficient data variability, as evidenced by a Sum of Squares value of 0.000 between groups and within groups, with degrees of freedom (df) of 1 and 10, respectively. This lack of variation meant that F-statistic and p-values could not be calculated, indicating a consistent absence or uniform presence of Cd across seasons.

In contrast, Copper concentrations showed a statistically significant difference across seasons, with an F-value of 5.640 and a p-value of 0.039. This led to rejecting the null hypothesis and suggesting that seasonal factors influence Cu levels in wetland ecosystems. Iron concentrations, however, did not exhibit significant seasonal variation ($F = 2.999$, $p = 0.114$), nor did Magnesium ($F = 0.495$, $p = 0.498$), Manganese ($F = 0.608$, $p = 0.454$), or Zinc ($F = 0.132$, $p = 0.724$). These findings indicate that, apart from Copper, the levels of the other metals remain relatively stable throughout the different seasons sampled. This stability may suggest consistent sources or environmental processes regulating these metals. In contrast, the variation in copper concentrations might be influenced by seasonal changes in factors such as runoff, agricultural activity, or hydrological dynamics that affect wetland ecosystems.

Table 4.21: ANOVA for metal concentrations by seasons

		Sum of Squares	Df	Mean Square	F	Sig.
Cd	Between Groups	0.000	1	0.000	.	.
	Within Groups	0.000	10	0.000		
	Total	0.000	11			
Cu	Between Groups	0.000	1	0.000	5.640	0.039
	Within Groups	0.000	10	0.000		
	Total	0.000	11			
Fe	Between Groups	0.001	1	0.001	2.999	0.114
	Within Groups	0.005	10	0.000		
	Total	0.006	11			
Mg	Between Groups	0.000	1	0.000	0.495	0.498
	Within Groups	0.000	10	0.000		
	Total	0.000	11			
Mn	Between Groups	0.000	1	0.000	0.608	0.454
	Within Groups	0.001	10	0.000		
	Total	0.001	11			
Zn	Between Groups	0.000	1	0.000	0.132	0.724
	Within Groups	0.001	10	0.000		
	Total	0.001	11			

b) ANOVA for metal concentrations by sampling stations

Similar analytical approaches and hypotheses were employed to those used for sampling levels in assessing the variation in metal concentrations across different sampling stations (Table 4.22). For Cadmium (Cd), the analysis was again limited by insufficient data variability, as indicated by a Sum of Squares value of 0.000, precluding the F-statistic and corresponding p-value calculation. This lack of detectable variation suggests consistently low or non-existent Cd concentrations across all stations sampled. Copper (Cu) concentrations showed no statistically significant differences across the stations, with an F-value of 0.747 and a p-value of 0.408, indicating that Cu distribution is relatively uniform throughout the study area. Similarly, Iron (Fe) concentrations did not significantly vary between sampling stations ($F = 0.212$, $p = 0.655$), further supporting the notion of a homogenous distribution pattern for this metal in the wetland ecosystems studied.

In contrast, Magnesium (Mg) concentrations exhibited significant variation across the sampling stations, with an F-value of 7.755 and a p-value of 0.019, leading to the rejection of the null hypothesis (H_0) of no difference. This suggests that Mg levels are influenced by location-specific factors, possibly related to localized geochemical characteristics, anthropogenic inputs, or hydrological conditions. Manganese (Mn) concentrations, on the other hand, did not show significant differences across the stations ($F = 1.894$, $p = 0.199$), nor did Zinc (Zn), which had an F-value of 0.130 and a p-value of 0.726.

Table 4.22: ANOVA for metal concentrations by sampling stations

		Sum of Squares	Df	Mean Square	F	Sig.
Cd	Between Groups	0.000	1	0.000	0.000	0.000
	Within Groups	0.000	10	0.000		
	Total	0.000	11			
Cu	Between Groups	0.000	1	0.000	0.747	0.408
	Within Groups	0.000	10	0.000		
	Total	0.000	11			
Fe	Between Groups	0.000	1	0.000	0.212	0.655
	Within Groups	0.006	10	0.001		
	Total	0.006	11			
Mg	Between Groups	0.000	1	0.000	7.755	0.019
	Within Groups	0.000	10	0.000		
	Total	0.000	11			
Mn	Between Groups	0.000	1	0.000	1.894	0.199
	Within Groups	0.001	10	0.000		
	Total	0.001	11			
Zn	Between Groups	0.000	1	0.000	0.130	0.726
	Within Groups	0.001	10	0.000		
	Total	0.001	11			

These results indicate that except for Magnesium, the concentrations of the analyzed metals remain relatively consistent across sampling stations within the wetland systems. The significant spatial variation in Mg underscores the importance of considering site-specific influences in environmental monitoring and management efforts. At the same time, the overall uniformity of the other metals suggests diffuse or widespread sources impacting the wetland ecosystems.

c) ANOVA for metal concentrations by sampling levels

An Analysis of Variance (ANOVA) was also performed to evaluate whether there were significant differences in metal concentrations across the different sampling levels, specifically downstream, midstream, and upstream locations within the study area (Table 4.23). However, for Cadmium (Cd), meaningful ANOVA analysis could not be conducted due to insufficient data variability, as indicated by a Sum of Squares value of zero, which meant that no F or p-values could be generated for this metal. This lack of variation in Cd concentrations suggests either extremely low or undetectable levels of Cd across all sampling points or insufficient sample representation for this particular metal.

The ANOVA results showed no statistically significant differences across the sampling levels for the remaining metals analyzed (Table 4.23). For example, copper (Cu) concentrations showed negligible variation between upstream, midstream, and downstream sites, with an F-value of 0.045 and a very high p-value of 0.956, indicating a strong likelihood that observed differences were due to random chance rather than systematic spatial variation. Iron (Fe) concentrations also demonstrated stability across the sampling gradient, with an F-value of 0.372 and a p-value of 0.699, further supporting the absence of significant spatial differences in Fe levels. Magnesium (Mg) concentrations exhibited a similar pattern, with no significant differences observed between sampling levels ($F = 0.180$, $p = 0.838$).

Manganese (Mn) and Zinc (Zn) concentrations also showed consistent levels regardless of sampling position, with Mn showing an F-value of 0.709 ($p = 0.518$) and Zn having a slightly higher F-value of 2.111 but still not reaching statistical significance with a p-value of 0.177. Collectively, these results indicate that the concentrations of these heavy metals are relatively uniform across different points along the wetland watercourses, suggesting either similar sources or adequate mixing

and dispersal mechanisms within the aquatic environment. This uniformity in metal levels across sampling locations implies that localized factors at each site may not significantly influence metal distribution at the scale assessed in this study. Consequently, environmental or anthropogenic inputs affecting metal concentrations may be widespread or diffuse rather than localized, highlighting the importance of considering the wetland ecosystem as an integrated unit when evaluating metal contamination and its potential ecological impacts.

Table 4.23: ANOVA for metal concentrations by sampling levels

		Sum of Squares	Df	Mean Square	F	Sig.
Cd	Between Groups	0.000	2	0.000	0.000	0.000
	Within Groups	0.000	9	0.000		
	Total	0.000	11			
Cu	Between Groups	0.000	2	0.000	0.045	0.956
	Within Groups	0.000	9	0.000		
	Total	0.000	11			
Fe	Between Groups	0.000	2	0.000	0.372	0.699
	Within Groups	0.006	9	0.001		
	Total	0.006	11			
Mg	Between Groups	0.000	2	0.000	0.180	0.838
	Within Groups	0.000	9	0.000		
	Total	0.000	11			
Mn	Between Groups	0.000	2	0.000	0.709	0.518
	Within Groups	0.001	9	0.000		
	Total	0.001	11			
Zn	Between Groups	0.000	2	0.000	2.111	0.177
	Within Groups	0.001	9	0.000		
	Total	0.001	11			

d) LSD for sampling levels

To further explore the differences in metal concentrations across the various sampling levels—namely downstream, midstream, and upstream—a post-hoc Least Significant Difference (LSD) test was conducted following the initial ANOVA analysis. The LSD test is a powerful statistical tool that allows for pairwise comparisons between group means, helping to identify specific differences that may not be apparent through ANOVA alone. In this study, the LSD results indicated that there were no statistically significant differences in the mean concentrations of the metals Copper (Cu), Iron (Fe), Magnesium (Mg), Manganese (Mn), and Zinc (Zn) between the three sampling levels (Table 4.24). This was evidenced by p-values exceeding the conventional significance threshold of 0.05 for all pairwise comparisons. These findings suggest a uniform distribution of these metals along the river continuum, from upstream through midstream to downstream locations within the wetland ecosystems. This uniformity implies that, within the scope of this study, metal concentrations do not significantly fluctuate based on the sampling position along the watercourse.

The results from the LSD test effectively confirm and support the earlier conclusions drawn from the ANOVA analyses. Specifically, the ANOVA revealed significant differences in metal concentrations related to seasonality and sampling stations but not across sampling levels. Copper concentrations were notably influenced by seasonal variation, highlighting that temporal environmental factors—such as changes in rainfall, temperature, and runoff—may impact Cu inflows and mobility within the wetlands. Similarly, Magnesium concentrations varied significantly depending on the sampling station, suggesting spatial heterogeneity likely influenced by localized environmental conditions, land use practices, or geological factors. Conversely, for other metals analyzed, including Cadmium (Cd), Iron (Fe), Manganese (Mn), and Zinc (Zn), the ANOVA results indicated no significant differences across seasons, stations, or sampling levels. This lack of variation suggests either a stable presence of these metals across the studied parameters or limitations in the sensitivity of the current sampling design to detect subtle spatial or temporal fluctuations.

The post-hoc LSD analyses serve as a valuable validation step, reinforcing the conclusion that the metals studied exhibit no significant concentration differences between upstream, midstream, and downstream sites. This finding points toward a

relatively homogenous distribution of these metals along the wetland watercourses, potentially due to consistent sources of inputs, effective natural mixing processes, or similar geochemical conditions across the sampling levels. However, the absence of significant differences should not be interpreted as conclusive evidence that no variations exist. Environmental systems are complex, and factors such as sample size, sampling frequency, and analytical sensitivity can influence the detection of spatial and temporal patterns.

Given the current results, it is recommended that future research efforts consider expanding the sample size and increasing the frequency of sampling events to enhance statistical power and sensitivity. Additionally, incorporating other environmental parameters and advanced analytical techniques might help detect more nuanced interactions or trends in metal concentrations not captured in this study. Understanding these subtleties is critical for effectively managing and conserving wetland ecosystems, particularly in regions experiencing diverse land use pressures and environmental changes.

In summary, the combination of ANOVA and post-hoc LSD testing in this study has provided valuable insights into the patterns of metal contamination within the wetland ecosystems of Murang'a County (Table 4.24). While significant seasonal and spatial variations were observed for some metals, Cu, Fe, Mg, Mn, and Zn concentrations were consistent across sampling levels, suggesting a degree of spatial homogeneity. These findings emphasize the importance of multi-faceted statistical approaches in environmental monitoring studies and highlight areas for further investigation to understand better the factors driving metal distribution and inflows in wetland environments.

Table 4.24: LSD for heavy metals' sampling levels

Dependent Variable	(I) Sampling level	(J) Sampling level	Mean Difference (I-J)	Std. Error	Sig.	Lower Bound	Upper Bound
Cu	Down-Stream	Mid-stream	0.0004061845	0.0019381604	0.839	-0.003978239	0.0047906078
Cu	Down-Stream	Up-Stream	0.0005634172	0.0019381604	0.778	-0.003821006	0.0049478405
Cu	Mid-stream	Down-Stream	-0.000406184	0.0019381604	0.839	-0.004790607	0.003978239
Cu	Mid-stream	Up-Stream	0.0001572327	0.0019381604	0.937	-0.004277094	0.004591560
Cu	Up-Stream	Down-Stream	-0.000563417	0.0019381604	0.778	-0.004947841	0.003821006
Cu	Up-Stream	Mid-stream	-0.000157233	0.0019381604	0.937	-0.004591561	0.004277094
Fe	Down-Stream	Mid-stream	-0.013044580	0.017498239	0.475	-0.052607526	0.026518973
Fe	Down-Stream	Up-Stream	0.000605686	0.017498239	0.998	-0.039512974	0.040724346
Fe	Mid-stream	Down-Stream	0.013044579	0.017498239	0.475	-0.026518973	0.052607526
Fe	Mid-stream	Up-Stream	0.013095283	0.017498239	0.473	-0.026463158	0.052653723
Fe	Up-Stream	Down-Stream	-0.00060569	0.017498239	0.998	-0.040724346	0.039512974

Fe	Up- Strea m	Mid- stream	- 0.013095 283	0.017498 239	0.4 73	- 0.052653 724	0.026463 148
Mg	Down- Strea m	Mid- stream	- 0.000285 677	0.001215 8204	0.8 26	- 0.002653 815	0.002082 462
Mg	Down- Strea m	Up- Strea m	0.000086 519	0.001215 8204	0.9 45	- 0.002663 815	0.002836 938
Mg	Mid- stream	Down- Strea m	0.000285 477	0.001215 8204	0.8 26	- 0.002082 462	0.002653 525
Mg	Mid- stream	Up- Strea m	0.000674 393	0.001215 8204	0.5 94	- 0.002078 203	0.003426 178
Mg	Up- Strea m	Down- Strea m	- 0.000086 519	0.001215 8204	0.9 45	- 0.002836 938	0.002663 815
Mg	Up- Strea m	Mid- stream	- 0.000674 393	0.001215 8204	0.5 94	- 0.003426 178	0.002078 207
Mn	Down- Strea m	Mid- stream	- 0.004136 582	0.005674 1471	0.4 85	- 0.016972 395	0.008699 239
Mn	Down- Strea m	Up- Strea m	0.002557 160	0.005674 1471	0.6 63	- 0.012078 652	0.017193 972
Mn	Mid- stream	Down- Strea m	0.004136 582	0.005674 1471	0.4 85	- 0.008699 239	0.016972 395
Mn	Mid- stream	Up- Strea m	- 0.006693 742	0.005674 1471	0.2 68	- 0.019592 255	0.006204 771
Mn	Up- Strea m	Down- Strea m	- 0.002557 160	0.005674 1471	0.6 63	- 0.017193 972	0.012079 652

Mn	Up-Stream	Mid-stream	0.006693 742	0.005674 1471	0.2 68	- 0.006204 771	0.019592 255
Zn	Down-Stream	Mid-stream	- 0.010666 209	0.007321 3291	0.1 79	- 0.027223 782	0.005891 364
Zn	Down-Stream	Up-Stream	0.003853 022	0.007321 3291	0.6 11	- 0.012708 075	0.020514 090
Zn	Mid-stream	Down-Stream	0.010666 208	0.007321 3291	0.1 79	- 0.005891 364	0.027223 782
Zn	Mid-stream	Up-Stream	0.014519 231	0.007321 3291	0.0 79	- 0.001631 288	0.030669 750
Zn	Up-Stream	Down-Stream	- 0.003853 022	0.007321 3291	0.6 11	- 0.020514 090	0.012708 075
Zn	Up-Stream	Mid-stream	- 0.014519 231	0.007321 3291	0.0 79	- 0.030102 282	0.002427 662

e) The hypothesis tested

Analysis of variance (ANOVA) was employed to determine whether significant differences exist in the concentrations of various metals across different temporal and spatial factors—namely, seasons, sampling stations, and sampling depths within wetland ecosystems. This approach allowed the study to evaluate how environmental conditions and land use variations influence the distribution and inflow of metals in these ecosystems. The wetland ecosystems in Murang'a County, characterized by diverse land use patterns, including agricultural activities, urban settlements, and natural vegetation, offer a complex matrix where metal concentrations may fluctuate due to natural processes and anthropogenic inputs. By applying ANOVA, the study was able to statistically test whether these fluctuations are significant, thereby providing insights into the dynamics of metal inflows and their potential ecological impacts.

The ANOVA results revealed significant seasonal and spatial variations in the concentrations of certain metals, particularly Copper (Cu) and Magnesium (Mg). Copper showed substantial differences in concentration between the dry and rainy seasons. This seasonal variation suggests that Cu levels in the wetlands are influenced by rainfall patterns, runoff intensity, and possibly seasonal agricultural practices like fertilizer and pesticide application. During the rainy season, increased runoff can mobilize copper from soils and anthropogenic sources into the wetlands, leading to higher concentrations. Conversely, during the dry season, reduced surface flow may limit copper transport, resulting in lower concentrations. This seasonal fluctuation underscores the dynamic nature of metal inflows and highlights the importance of considering temporal variability when assessing metal pollution in wetland ecosystems.

Magnesium concentrations exhibited significant differences across sampling stations, indicating that localized environmental or land use conditions within Murang'a County influence the spatial distribution of Mg. This spatial heterogeneity could be attributed to variations in geological substrates, soil types, or anthropogenic inputs such as fertilizer use in agricultural zones or effluent discharge from nearby settlements. The finding suggests that certain areas within the wetlands receive higher magnesium loads than others, which may impact the aquatic ecosystems' biogeochemical processes and overall health differently depending on the location. This spatial variation reinforces the need for site-specific monitoring and management strategies rather than a one-size-fits-all approach to wetland conservation and pollution control.

In contrast, metals such as Iron (Fe), Manganese (Mn), and Zinc (Zn) did not show statistically significant differences across seasons, sampling stations, or sampling depths. This lack of substantial variation suggests that these metals have a more uniform distribution within the wetlands or that their concentrations are less sensitive to the environmental factors and land use practices considered in this study. Their consistent levels might be due to their natural abundance in the region's geological formations or stable inputs from diffuse sources. Alternatively, it could indicate that Fe, Mn, and Zn are more tightly bound within sediments or subject to biological regulation that buffers their concentrations against environmental fluctuations. This

pattern highlights the complex interactions between metals and their environment and suggests that not all heavy metals respond similarly to environmental changes or land use changes. These findings support the hypothesis that there are measurable differences in the inflows of nutrients and heavy metals in the small wetland ecosystems of Murang'a County. Understanding these differences is crucial for developing effective environmental management practices, as contaminants' spatial and temporal variability affects the health of wetland biota and the services these ecosystems provide.

To complement the ANOVA findings, Pearson correlation analysis explored potential relationships between environmental or land use variables (such as agricultural intensity, proximity to urban areas, or vegetation cover) and the concentrations of nutrients and heavy metals in the wetlands. While the average correlation coefficients were generally low, indicating weak to moderate overall linear relationships, the data showed a wide range of correlation values. This variability suggests that certain land use practices or environmental factors have substantial localized impacts on specific metal concentrations—either increasing or decreasing them significantly. For example, areas with intensive agriculture might show strong positive correlations with metals like copper or magnesium due to agrochemicals containing these elements. Conversely, urban runoff could negatively correlate with certain metals if mitigation measures are in place or dilution effects dominate. Both positive and negative strong correlations underscore the heterogeneous nature of metal inflows and the complex interplay between anthropogenic activities and natural processes.

Overall, the combined use of ANOVA and Pearson correlation analysis demonstrates that environmental and land use factors significantly influence the distribution and inflow of heavy metals in wetland ecosystems. These findings highlight the need for targeted monitoring and management interventions that consider both temporal changes and spatial differences to safeguard the ecological integrity of wetlands in Murang'a County effectively.

4.7.3 Pollution Index (PI) for Heavy Metals

Pollution index for each of the detected heavy metals was calculated where Lead (Pb) was found to have high pollution with **PI > 5** thus, raising urgent concern. All other metals (Fe, Cd, Zn, Mn, Cu, Mg) were well below the threshold. This research findings

therefore conclude that Lead contamination is significantly above the safe threshold and poses a high environmental and health risk in the Murang'a Wetlands. Table 4.25 shows calculated pollution indexes for each of the analysed metals in the water samples from Murang'a wetlands.

Table 4.25: Pollution Index (PI) for Heavy Metals

Element	C _i	S _i	PI = C _i /S _i	Pollution Level
Iron (Fe)	0.0246	0.3	0.082	Low
Lead (Pb)	0.2623	0.01	26.23	High
Cadmium (Cd)	0.0000	0.003	0.000	Low
Zinc (Zn)	0.0198	3.0	0.0066	Low
Manganese (Mn)	0.0055	0.1	0.055	Low
Copper (Cu)	0.0028	2.0	0.0014	Low
Magnesium (Mg)	0.0019	50	0.000038	Low

4.8 Pesticide Inflows in Murang'a Wetland Ecosystems

The pesticide inflows were analyzed using water and sediment samples collected from the Maragua and Mathioya wetland ecosystems. Four pesticides—Malathion, Carbendazim, Thiamethoxam, and Diazinon—were targeted for detection and quantification. The results indicated that Malathion, Thiamethoxam, and Diazinon were present in varying concentrations across the dry and rainy seasons, reflecting seasonal fluctuations and differing contamination levels in the two ecosystems. In contrast, Carbendazim was not detected in any water or sediment samples throughout the study period, with its concentration consistently recorded as zero. These findings are summarized in Table 4.26, which presents the detailed concentration levels of each pesticide across the sampling sites and seasons.

Table 4.26: Pesticide inflow analysis in water and sediment samples

Sam ple ID	Stream	Samp ling statio n	Seas on	Samp le type	Samplin g Code	Malathi on Conc.(mg/L)	Thiameth oxam Conc.(m g/L)	Diazino n Conc.(mg/L)	Carbend azim Conc.(mg/L)
1b	Upstrea m	Mara gua	Wet	Water	MAR/W W/A2	0.0188	<LOQ	0.0018	<LOQ
2b	Middlest ream	Mara gua	Wet	Water	MAR/W W/B2	0.0038	<LOQ	0.0005	<LOQ
3b	Lowerst ream	Mara gua	Wet	Water	MAR/W W/C2	0.0150	<LOQ	0.0047	<LOQ
4b	Upstrea m	Mathi oya	Wet	Water	MAT/W W/A2	0.0051	<LOQ	0.0015	<LOQ
5b	Middlest ream	Mathi oya	Wet	Water	MAT/W W/B2	0.0026	<LOQ	0.0009	<LOQ
6b	Lowerst ream	Mathi oya	Wet	Water	MAT/W W/C2	0.0045	<LOQ	<LOQ	<LOQ
7b	Upstrea m	Mara gua	Dry	Water	MAR/W D/A2	0.0045	<LOQ	0.0014	<LOQ
8b	Middlest ream	Mara gua	Dry	Water	MAR/W D/B2	0.0094	0.0002	0.0042	<LOQ
9b	Lowerst ream	Mara gua	Dry	Water	MAR/W D/C2	0.0041	<LOQ	<LOQ	<LOQ
10b	Upstrea m	Mathi oya	Dry	Water	MAT/W D/A2	0.0091	0.0006	0.0030	<LOQ
11b	Middlest ream	Mathi oya	Dry	Water	MAT/W D/B2	0.0035	<LOQ	0.0010	<LOQ
12b	Lowerst ream	Mathi oya	Dry	Water	MAT/W D/C2	0.0041	<LOQ	0.0013	<LOQ
1s	Upstrea m	Mara gua	Wet	Sedi ment	MAR/S W/A2	0.0051	0.0040	0.0015	<LOQ
2s	Middlest ream	Mara gua	Wet	Sedi ment	MAR/S W/B2	0.0048	0.0027	<LOQ	<LOQ
3s	Lowerst ream	Mara gua	Wet	Sedi ment	MAR/S W/C2	0.0043	0.0011	0.0014	<LOQ
4s	Upstrea m	Mathi oya	Wet	Sedi ment	MAT/S W/A2	0.0007	0.0044	0.0004	<LOQ
5s	Middlest ream	Mathi oya	Wet	Sedi ment	MAT/S W/B2	0.0012	<LOQ	<LOQ	<LOQ
6s	Lowerst ream	Mathi oya	Wet	Sedi ment	MAT/S W/C2	0.0021	0.0003	<LOQ	<LOQ
7s	Upstrea m	Mara gua	Dry	Sedi ment	MAR/S D/A2	0.0048	0.0015	<LOQ	<LOQ
8s	Middlest ream	Mara gua	Dry	Sedi ment	MAR/S D/B2	0.0105	0.0015	0.0025	<LOQ
9s	Lowerst ream	Mara gua	Dry	Sedi ment	MAR/S D/C2	0.0038	0.0003	0.0013	<LOQ
10s	Upstrea m	Mathi oya	Dry	Sedi ment	MAT/SD /A2	0.0047	0.0029	0.0012	<LOQ
11s	Middlest ream	Mathi oya	Dry	Sedi ment	MAT/SD /B2	0.0063	0.0020	0.0015	<LOQ
12s	Lowerst ream	Mathi oya	Dry	Sedi ment	MAT/SD /C2	0.0055	0.0048	0.0025	<LOQ

4.8.1 Descriptive analysis of pesticide inflows and sediment samples

The analysis of pesticide concentrations in small wetland ecosystems in Murang'a County revealed varying distribution patterns and potential contamination risks associated with different pesticides. The descriptive statistics for Malathion, Thiomethoxam, and Diazinon concentrations provide insights into these chemicals' central tendencies, dispersion, and distribution shapes in the environment, while Carbendazim concentrations remain undetected. Malathion concentration levels across the 24 samples showed considerable variation, ranging from 0.00068941 mg/L to 0.01876634 mg/L. The average concentration of Malathion was 0.00575524 mg/L, indicating a moderate presence of this pesticide in the wetland samples (Figure 4.18). The positive skewness (1.856) suggests that the distribution of Malathion concentrations is asymmetrically distributed with a right tail, implying that a few samples contain significantly higher concentrations than the average. The kurtosis value of 3.725 indicates a leptokurtic distribution characterized by a sharper peak and more pronounced outliers than a normal distribution. These statistical indicators suggest that Malathion levels could occasionally spike in certain areas or under specific conditions, potentially posing a risk to the wetland ecosystem.

Thiomethoxam concentrations, on the other hand, exhibit a range from -0.0023556 mg/L to 0.0047881 mg/L, with a mean concentration of 0.000580075 mg/L. The negative minimum value might indicate measurement errors or data inaccuracies, which should be addressed before drawing definitive conclusions. The distribution of Thiomethoxam is slightly right-skewed (skewness of 0.656) and has a negative kurtosis value of -0.391, indicating a flatter distribution with fewer extreme values than a normal distribution. This suggests Thiomethoxam is generally present in lower concentrations, with less variability than Malathion.

Diazinon concentrations also vary, ranging from -0.00049465 mg/L to 0.0046815 mg/L, with an average concentration of 0.001241848 mg/L. Similar to Thiomethoxam, the negative minimum value raises concerns about the accuracy of the data, which may need further review. The distribution of Diazinon concentrations is moderately right-skewed (skewness of 0.786), indicating a tendency for higher values in a few samples. In contrast, the kurtosis value of 0.565 suggests a slightly

peaked distribution. This pattern reflects the potential for Diazinon to be present in significant concentrations in certain areas, though it remains relatively low on average.

Finally, Carbendazim concentrations were consistently measured at 0.0000 mg/L across all samples, indicating that this pesticide was either not detected or absent from the study area's wetland ecosystems. The lack of variation in Carbendazim concentrations suggests that it does not contribute to pesticide contamination in these wetlands, contrasting sharply with the presence of Malathion, Thiomethoxam, and Diazinon.

Descriptive Statistics												
	N	Range	Minimum	Maximum	Mean		Std. Deviation	Variance	Skewness		Kurtosis	
	Statistic	Statistic	Statistic	Statistic	Statistic	Std. Error	Statistic	Statistic	Statistic	Std. Error	Statistic	Std. Error
Malathion Conc.(mg/L)	24.0000	.0181	.0007	.0188	.0058	.0009	.0042	.0000	1.8562	.4723	3.7249	.9178
Thiomethoxam Conc. (mg/L)	24.0000	.0071	-.0024	.0048	.0006	.0004	.0020	.0000	.6556	.4723	-.3906	.9178
Diazinon Conc.(mg/L)	24.0000	.0052	-.0005	.0047	.0012	.0003	.0014	.0000	.7864	.4723	.5647	.9178
Carbendazim Conc. (mg/L)	24.0000	.0000	.0000	.0000	.0000	.0000	.0000	.0000	.	.	.	.
Valid N (listwise)	24.0000											

Figure 4.18: Descriptive statistics for pesticides, from water and sediment samples

4.8.2 Correlation analysis on pesticide inflows

The correlation analysis among the pesticide concentrations in the wetland ecosystems of Murang'a County provides insight into the relationships between different pesticides (Figure 4.19). The key findings from the Pearson correlation coefficients are as follows: There is a strong positive correlation between Malathion and Diazinon concentrations, with a Pearson correlation coefficient of 0.6697. This correlation is statistically significant at the 0.01 level ($p = 0.0003$), indicating that as the concentration of Malathion increases, the concentration of Diazinon also tends to increase. This strong association suggests that these two pesticides may co-occur in the environment due to similar sources or application practices. The correlation between Malathion and Thiomethoxam concentrations is weak and negative, with a Pearson correlation coefficient of -0.2423. However, this correlation is not statistically

significant ($p = 0.2540$), implying no meaningful linear relationship between these two pesticides. The weak negative correlation might indicate that they are applied independently of each other or have different environmental behaviors.

The correlation between Thiamethoxam and Diazinon concentrations is almost nonexistent, with a Pearson correlation coefficient of -0.0025 , and this relationship is not statistically significant ($p = 0.9909$). This result suggests no association between the concentrations of these two pesticides in the wetland samples. The correlation coefficients for Carbendazim concentrations could not be computed because this variable is constant across all samples (i.e., all values are zero). Since there is no variation in Carbendazim concentrations, it cannot be correlated with other pesticide concentrations. The significant positive correlation between Malathion and Diazinon suggests a possible link in their environmental presence, potentially due to similar sources or usage patterns. In contrast, Thiomethoxam shows no significant correlation with Malathion or Diazinon, indicating that its distribution is independent of the other two pesticides. The absence of variability in Carbendazim concentrations prevents any meaningful correlation analysis with this pesticide.

Correlations					
		Malathion Conc.(mg/L)	Thiomethoxa m Conc. (mg/L)	Diazinon Conc.(mg/L)	Carbendazim Conc.(mg/L)
Malathion Conc.(mg/L)	Pearson Correlation	1.0000	-.2423	.6697**	. ^b
	Sig. (2-tailed)		.2540	.0003	.
	N	24.0000	24.0000	24.0000	24.0000
Thiomethoxam Conc. (mg/L)	Pearson Correlation	-.2423	1.0000	-.0025	. ^b
	Sig. (2-tailed)	.2540		.9909	.
	N	24.0000	24.0000	24.0000	24.0000
Diazinon Conc.(mg/L)	Pearson Correlation	.6697**	-.0025	1.0000	. ^b
	Sig. (2-tailed)	.0003	.9909		.
	N	24.0000	24.0000	24.0000	24.0000
Carbendazim Conc. (mg/L)	Pearson Correlation	. ^b	. ^b	. ^b	. ^b
	Sig. (2-tailed)	.	.	.	.
	N	24.0000	24.0000	24.0000	24.0000

** . Correlation is significant at the 0.01 level (2-tailed).

b . Cannot be computed because at least one of the variables is constant.

Figure 4.19: Correlation analysis for pesticide inflows

4.8.3 ANOVA on pesticide inflows

i. ANOVA by streams

The analysis of variance (ANOVA) for pesticide concentrations across different stream locations in the wetlands of Murang'a County reveals several key insights (Figure 4.20). Malathion's ANOVA results indicate no statistically significant difference in concentration among the lower stream, middle stream, and upstream locations. The p-value of 0.791, well above the significance level of 0.05, suggests that the Malathion levels are relatively consistent across these different locations, implying that the position along the stream does not influence its distribution in the wetland environment.

Similarly, for Thiomethoxam, the ANOVA results show no significant variation in concentration across the different stream locations, with a p-value of 0.733. This finding suggests that Thiomethoxam is evenly distributed along the streams, and its presence is likely uniform regardless of the specific stream location. This uniformity may reflect similar environmental or agricultural practices affecting all stream sections equally. Diazinon also shows no significant differences in concentration between the stream locations, as indicated by a high p-value of 0.984. This result reinforces the pattern observed with Malathion and Thiomethoxam, where the pesticide appears evenly distributed along the streams. The lack of variation suggests that Diazinon, like the other pesticides, may be introduced into the environment through sources that affect the entire stream rather than specific sections (Figure 4.20).

The ANOVA could not be performed for Carbendazim because its concentration was constant (0.000 mg/L) across all samples. This constancy indicates that Carbendazim was either absent or below detectable levels in all the locations sampled. As a result, no significant variation could be observed or analyzed for this pesticide.

ANOVA						
		Sum of Squares	df	Mean Square	F	Sig.
Malathion Conc.(mg/L)	Between Groups	.0000	2.0000	.0000	.2373	.7909
	Within Groups	.0004	21.0000	.0000		
	Total	.0004	23.0000			
Thiomethoxam Conc. (mg/L)	Between Groups	.0000	2.0000	.0000	.3149	.7332
	Within Groups	.0001	21.0000	.0000		
	Total	.0001	23.0000			
Diazinon Conc.(mg/L)	Between Groups	.0000	2.0000	.0000	.0160	.9842
	Within Groups	.0000	21.0000	.0000		
	Total	.0000	23.0000			
Carbendazim Conc. (mg/L)	Between Groups	.0000	2.0000	.0000		
	Within Groups	.0000	21.0000	.0000		
	Total	.0000	23.0000			

Figure 4.20: ANOVA on pesticide inflows

The post hoc analysis using the Least Significant Difference (LSD) method further supports these findings. No significant differences existed in the concentrations of Malathion, Thiomethoxam, and Diazinon between any stream locations. All pairwise comparisons returned p-values greater than 0.05, confirming that the mean differences in pesticide concentrations between the lower stream, middle stream, and upstream are not statistically significant.

Multiple Comparisons							
LSD							
Dependent Variable	(I) Streams	(J) Streams	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
						Lower Bound	Upper Bound
Malathion Conc.(mg/L)	lowerstream	middlestream	.0002	.0022	.9417	-.0043	.0046
		upstream	-.0012	.0022	.5840	-.0057	.0033
	middlestream	lowerstream	-.0002	.0022	.9417	-.0046	.0043
		upstream	-.0014	.0022	.5354	-.0058	.0031
	upstream	lowerstream	.0012	.0022	.5840	-.0033	.0057
		middlestream	.0014	.0022	.5354	-.0031	.0058
Thiomethoxam Conc. (mg/L)	lowerstream	middlestream	.0001	.0010	.9278	-.0021	.0023
		upstream	-.0007	.0010	.5311	-.0028	.0015
	middlestream	lowerstream	-.0001	.0010	.9278	-.0023	.0021
		upstream	-.0008	.0010	.4743	-.0029	.0014
	upstream	lowerstream	.0007	.0010	.5311	-.0015	.0028
		middlestream	.0008	.0010	.4743	-.0014	.0029
Diazinon Conc.(mg/L)	lowerstream	middlestream	.0000	.0007	.9893	-.0015	.0015
		upstream	-.0001	.0007	.8736	-.0017	.0014
	middlestream	lowerstream	.0000	.0007	.9893	-.0015	.0015
		upstream	-.0001	.0007	.8842	-.0016	.0014
	upstream	lowerstream	.0001	.0007	.8736	-.0014	.0017
		middlestream	.0001	.0007	.8842	-.0014	.0016

Figure 4.21: LSD on pesticide inflows

In conclusion, the results from the ANOVA and post hoc analyses (Figure 4.21) indicate that Malathion, Thiomethoxam, and Diazinon concentrations are uniformly distributed across the different stream locations in the wetlands of Murang'a County. Carbendazim's constant and undetectable levels suggest that it does not vary between locations. These findings imply that the factors contributing to pesticide presence in these wetlands are consistent across the various stream sections, possibly due to widespread agricultural practices or environmental conditions that affect the entire watershed.

ii. ANOVA by sampling stations

The ANOVA results revealed a marginally significant difference in Malathion concentration between the two sampling stations, with a p-value of 0.051 (Figure 4.22). This suggests that there may be slight variations in Malathion levels between the stations, but this difference is only borderline significant, meaning it might be due to chance. The between-group sum of squares is 0.0001, and the within-group sum of squares is 0.0003, indicating that most of the variation is within the stations rather than between them. For Thiomethoxam, the ANOVA shows no significant difference between the stations, with a p-value of 0.631. This indicates that the concentration of Thiamethoxam is consistent across the different sampling stations, suggesting that the source or factors influencing its presence are similar in both locations. The analysis also indicates no significant difference in Diazinon concentration between the stations (p-value = 0.370). Like Thiomethoxam, this suggests that Diazinon levels are uniform across the stations, with no notable spatial variability. Like previous analyses, Carbendazim shows no variability between the stations, as indicated by the absence of a computed F statistic. This is due to its concentration being constant (or undetectable) across all samples.

		ANOVA				
		Sum of Squares	df	Mean Square	F	Sig.
Malathion Conc.(mg/L)	Between Groups	.0001	1.0000	.0001	4.2595	.0510
	Within Groups	.0003	22.0000	.0000		
	Total	.0004	23.0000			
Thiomethoxam Conc. (mg/L)	Between Groups	.0000	1.0000	.0000	.2378	.6306
	Within Groups	.0001	22.0000	.0000		
	Total	.0001	23.0000			
Diazinon Conc.(mg/L)	Between Groups	.0000	1.0000	.0000	.8392	.3696
	Within Groups	.0000	22.0000	.0000		
	Total	.0000	23.0000			
Carbendazim Conc. (mg/L)	Between Groups	.0000	1.0000	.0000	.	.
	Within Groups	.0000	22.0000	.0000		
	Total	.0000	23.0000			

Figure 4.22: ANOVA on pesticide inflows by sampling stations

iii. ANOVA by Seasons

The seasonal analysis reveals no significant difference in Malathion levels between the dry and wet seasons, with a p-value of 0.916 (Figure 4.23). This indicates that Malathion concentrations are stable across different seasons, possibly reflecting consistent pesticide application or environmental conditions throughout the year. Similarly, the ANOVA for Thiomethoxam concentration shows no significant seasonal variation (p-value = 0.693). The consistent levels suggest that Thiomethoxam, like Malathion, is not significantly affected by seasonal changes, indicating that its presence in the wetlands may be due to continuous or widespread use throughout the year. Although the ANOVA shows a somewhat lower p-value of 0.223 for Diazinon, it still indicates no significant seasonal difference. The lack of seasonal variation in Diazinon concentration suggests that seasonal changes do not influence the environmental or anthropogenic factors contributing to its presence. Once again, no significant difference is observed in Carbendazim concentration between the seasons due to its constant concentration across all samples.

ANOVA						
		Sum of Squares	Df	Mean Square	F	Sig.
Malathion Conc.(mg/L)	Between Groups	0.0000	1.0000	0.0000	0.0115	0.9157
	Within Groups	0.0004	22.0000	0.0000		
	Total	0.0004	23.0000			
Thiamethoxam Conc.(mg/L)	Between Groups	0.0000	1.0000	0.0000	0.1600	0.6930
	Within Groups	0.0001	22.0000	0.0000		
	Total	0.0001	23.0000			
Diazinon Conc.(mg/L)	Between Groups	0.0000	1.0000	0.0000	1.5704	0.2233
	Within Groups	0.0000	22.0000	0.0000		
	Total	0.0000	23.0000			
Carbendazim Conc.(mg/L)	Between Groups	0.0000	1.0000	0.0000	.	.
	Within Groups	0.0000	22.0000	0.0000		
	Total	0.0000	23.0000			

Figure 4.23: ANOVA on pesticides by seasons

The ANOVA results suggest that there are no significant differences in concentrations between different sampling stations or seasons for most of the pesticides analyzed. Malathion is the only pesticide that shows a borderline significant difference between stations, which might warrant further investigation. The lack of substantial seasonal variation implies that the factors influencing pesticide levels in these wetlands are consistent throughout the year, possibly reflecting steady agricultural practices or persistent environmental conditions in Murang'a County.

iv. ANOVA by sample type (sediment or water)

The analysis of variance (ANOVA) conducted to examine the differences in pesticide concentrations based on the type of sample (sediment or water) reveals distinct patterns for each pesticide (Table 4.27). The ANOVA results for Malathion indicate a p-value of 0.135, which suggests no significant difference in Malathion concentrations between sediment and water samples. The sum of squares between groups is 0.000, and within groups is 0.000, indicating minimal variability in Malathion levels regardless of the sample type. Thus, Malathion's presence is relatively consistent across different wetland sample types. For Thiometoxam, the ANOVA shows a significant difference between sample types, with a p-value of 0.000. This result indicates that Thiometoxam concentrations vary significantly between sediment and

water samples. The sum of squares between groups is 0.000, while the within-groups sum of squares is also 0.000, highlighting a substantial variation in Thiomethoxam levels based on sample type. This suggests that Thiomethoxam may be more prevalent in one sample type than the other, necessitating further investigation to understand the distribution pattern.

The ANOVA results for Diazinon reveal a p-value of 0.217, indicating no significant difference in Diazinon concentrations between sediment and water samples. Like Malathion, the sum of squares between groups and within groups is 0.000, suggesting that Diazinon levels are consistent across the sample types, with no notable variation. The ANOVA could not be performed for Carbendazim due to a lack of variability (constant concentration) across all samples. As Carbendazim was consistently undetectable or at zero concentration, there are no differences to report between sample types.

Table 4.27: ANOVA by sample type (water/sediment)

		Sum of Squares	Df	Mean Square	F	Sig.
Malathion Conc.(mg/L)	Between Groups	0.0000	1.000	0.0000	2.4028	0.1354
	Within Groups	0.0004	22.000	0.0000		
	Total	0.0004	23.000			
Thiamethoxam Conc.(mg/L)	Between Groups	0.0001	1.000	0.0001	25.1803	0.0001
	Within Groups	0.0000	22.000	0.0000		
	Total	0.0001	23.000			
Diazinon Conc.(mg/L)	Between Groups	0.0000	1.000	0.0000	1.6173	0.2168
	Within Groups	0.0000	22.000	0.0000		
	Total	0.0000	23.000			
Carbendazim Conc.(mg/L)	Between Groups	0.0000	1.000	0.0000	0.0000	0.0000
	Within Groups	0.0000	22.000	0.0000		
	Total	0.0000	23.000			

The ANOVA results show that Thiomethoxam is the only pesticide with a significant variation between sediment and water samples, suggesting differences in how Thiomethoxam distributes between these sample types. Malathion and Diazinon do not exhibit substantial differences based on sample type, indicating uniform distribution across sediments and water. The constant levels of Carbendazim across all samples mean no comparison could be made for this pesticide. These findings highlight the need for further analysis to understand the factors influencing Thiomethoxam distribution and to confirm the uniformity of Malathion and Diazinon concentrations.

4.9 Changes in water quality parameters due to catchment land use practices

Objective three states that we need to determine changes in water quality parameters concerning small wetlands catchment land-use systems with the hypothesis that land-use systems have no effects on water quality parameters (pH, EC, TDS, salinity, and turbidity). The hypothesis H_{03} posits that land use systems do not affect water quality parameters. To evaluate this hypothesis, we must analyze how various land use systems (agricultural, urban, and natural landscapes) influence key water quality indicators. These indicators include potential hydrogen concentration (pH), which indicates the acidity or alkalinity of water. Changes in pH can affect aquatic life and chemical processes in the water. Significant variation in pH across different land use systems could indicate impacts from agricultural runoff (which might introduce acidic or alkaline substances), urban runoff (which might introduce pollutants affecting the pH), or natural buffering capacities in natural landscapes. Electrical Conductivity (EC) measures the water's ability to conduct electricity, which is influenced by the water's concentration of dissolved salts (ions). Total Dissolved Solids (TDS) is the combined content of all inorganic and organic substances contained in a liquid. High TDS levels can indicate poor water quality. Higher values in agricultural and urban areas might indicate higher ion concentrations due to fertilizers, pesticides, and urban pollutants. Turbidity measures the clarity of water, affected by the presence of suspended particles. High turbidity can reduce light penetration and affect aquatic plants and animals' metabolic and physiological processes. Higher turbidity in agricultural and urban areas could result from soil erosion, construction activities, and other disturbances compared to more stable natural regions.

A comprehensive study involving data collection from various catchment areas with different land use systems is essential to determine changes in these water quality parameters. The analysis included sampling locations where the selected multiple locations represent different land use systems (agricultural, urban, natural). Temporal sampling was conducted to capture temporal variations, possibly seasonally. Table 4.28 presents the result of the water quality parameters analyzed in the water samples during both the wet and dry seasons. The statistical tests in this category are descriptive statistics that calculate the mean, standard deviation, range, skewness, and kurtosis for each water quality parameter. ANOVA is used to compare the means of water quality parameters across different land use systems and identify any

statistically significant differences. Post Hoc Tests (e.g., LSD) given ANOVA show substantial differences, so post hoc tests are conducted to pinpoint specific group differences. Correlation analysis assesses relationships between water quality parameters and land use types to identify potential influences and interactions.

Table 4.28: Water Quality Parameters

Wet land	Samp ling Level	Salinity (mg/L)		TDS (mg/L)		EC (mS/cm)		pH		Turbidity (NTU)	
		Wet	Dry	Wet	Dry	Wet	Dry	W et	Dr y	Wet	Dry
Mar agua	Upstream	71.7	71.7	64.9	65.4	0.098	0.096	5.5	5.81	177.5	140.5
	Midstream	161.6	159.6	150.6	148.3	0.228	0.219	6.01	6.37	183.8	158.4
	Downstream	141.4	150.2	139.3	135.9	0.199	0.203	6.4	6.45	128.3	166.9
Math ioya											
	Upstream	35.2	28.9	31.7	26.7	0.045	0.039	6.22	6.48	108.2	122.2
	Midstream	135	147.5	122.7	132.9	0.187	0.199	6.42	6.45	190.9	161.0
	Downstream	146	146.6	133.9	132.6	0.202	0.194	6.51	6.51	194.6	192.2

The water quality parameters varied across the sampling levels, as shown in Table 4.28. The general trend is that, upstream, the quality indicator values are low, but as we progressed downstream, there was an increase in the levels. This is attributed to intensive agricultural practices downstream along the riverine wetland ecosystems, which involve massive application of inorganic fertilizers, which, through runoff, get deposited in the water body as residue inflows. Such fertilizers contribute dissolved salts (ions) that facilitate the electrical conductivity of the water. Also, some

agrochemicals such as lime (alkaline), phosphatic, and nitrogenous fertilizers are acidic, thus influencing water p^H . Siltation in riverine wetlands contributes to variations in the turbidity of the water.

4.9.1 Descriptive Statistics for Water Quality Parameters

i. pH

The pH of water is a critical parameter that indicates its acidity or alkalinity, which in turn influences many biological, chemical, and physical processes within aquatic ecosystems. The pH scale ranges from 0 to 14, where values below 7 indicate acidic conditions, around seven are neutral, and above 7 indicate alkalinity. The pH level affects nutrient availability, metal solubility, and the overall health of aquatic organisms. In the current analysis, the pH values across the twelve wetland water samples ranged from 5.50 to 6.51, with an average pH of 6.26 ± 0.09 . This indicates that the wetland ecosystem exhibits slightly acidic conditions overall. Slight acidity in wetland waters is typical and can arise from natural and anthropogenic factors. One natural contributor to acidity is the decomposition of organic matter, which releases organic acids such as humic and fulvic acids into the water. Additionally, agricultural runoff, particularly from using inorganic phosphate fertilizers, can introduce acidic compounds that lower the pH. These inflows may lead to localized decreases in pH depending on the timing and intensity of runoff events.

The standard deviation of 0.32 indicates moderate variability in pH levels across the samples, meaning that while some fluctuations exist, most values remain relatively close to the mean pH. This suggests that the wetland environment maintains a somewhat stable acidic condition, with limited extremes in pH under normal circumstances. The skewness value of -1.57 ± 0.64 reflects a pronounced left-skewed distribution. This negative skewness means that more pH values are clustered below the mean, pulling the average pH towards the lower, more acidic end of the spectrum. Such a distribution suggests that episodic or localized acidifying factors are relatively common in the wetland, possibly linked to varying inputs of acidic substances from runoff or internal wetland processes such as organic matter breakdown.

Further, the kurtosis value of 1.68 ± 1.23 indicates a leptokurtic distribution, which is characterized by heavier tails and a sharper peak compared to a normal distribution.

This implies that the pH dataset contains a higher frequency of extreme values—lower and potentially higher—than expected under a normal distribution. These extreme values could represent periodic influxes of acidic substances, such as heavy rainfall events flushing acidic fertilizers or organic acids into the wetland or occasional biochemical processes that transiently alter the water's pH.

Thoughtfully, the pH variability and distribution are essential for assessing the ecological health of the wetland. Slightly acidic conditions may influence nutrient cycling and metal availability, potentially affecting sensitive aquatic organisms and plant species adapted to neutral or alkaline conditions. Additionally, sudden shifts in pH due to extreme values could stress biota and alter ecosystem functions. The pH analysis reveals a slightly acidic wetland environment with moderate variability and a tendency towards lower pH values. The leptokurtic distribution suggests occasional episodes of greater acidity that environmental or anthropogenic factors may drive. Continuous pH monitoring and investigation of runoff sources are recommended to understand the drivers of acidity better and develop management strategies to protect wetland health.

ii. Electrical Conductivity (EC)

Electrical Conductivity (EC) is a fundamental water quality parameter that measures the water's ability to conduct an electric current. This conductivity directly correlates with the concentration of dissolved ions, primarily salts in the water, such as sodium, chloride, calcium, and magnesium. Because dissolved salts increase the water's ionic strength, EC is an effective proxy for assessing the salinity or mineralization of water bodies. Monitoring EC is crucial in understanding the health of aquatic ecosystems, irrigation suitability, and the influence of anthropogenic activities on water quality.

In the current study, EC values measured across twelve wetland water samples ranged from 0.04 mS/cm to 0.23 mS/cm, with a mean of 0.16 ± 0.02 mS/cm. This average value suggests a moderate salinity level in the wetland system. Although this range of EC values is relatively low compared to seawater (~50 mS/cm) or highly saline waters, it still reflects a degree of dissolved salts that can influence aquatic life and vegetation health within the wetland.

The observed salinity level may result from several sources. Agricultural runoff is a prominent contributor, as fertilizers and soil amendments dissolve into surface runoff and groundwater, elevating the concentration of ions. Other anthropogenic activities such as wastewater discharge, urban runoff, or small-scale industrial inputs could also influence the EC levels. Natural factors such as mineral weathering in the catchment area and evapotranspiration in the wetland can further concentrate dissolved salts and thus impact the EC readings. The standard deviation of 0.07 mS/cm highlights moderate variability in EC values across the sampled sites. This variability indicates that salinity levels fluctuate spatially, possibly due to differing land use practices around the wetland, varying hydrological inputs, or seasonal changes affecting dilution and concentration processes. Areas receiving direct runoff from agricultural fields or urban landscapes may show higher EC values, while locations with more inflow of fresh water or better flushing might record lower salinity.

The statistical distribution of the EC data reveals further insights into the nature of the salinity variations. The skewness of -0.93 ± 0.64 indicates a left-skewed distribution, where a larger proportion of the values cluster toward the higher end of the EC range. This pattern suggests elevated salinity levels are more commonly observed across the wetland samples. Persistent sources such as continuous agricultural runoff or mineral dissolution could explain this concentration of higher EC values.

The kurtosis value of -0.87 ± 1.23 suggests a platykurtic distribution, indicating that the EC data are relatively evenly spread with fewer extremely high or low values than a normal distribution. This relatively flat distribution implies that while salinity varies across the wetland, extreme spikes or dips in conductivity are less frequent. Such a distribution pattern can result from stable environmental conditions or a balance of input sources that prevent abrupt changes in ionic concentrations. Understanding the EC levels and their statistical characteristics is vital for wetland management. Moderate salinity levels, if persistent or increasing, may affect wetland vegetation composition, soil chemistry, and aquatic organisms, some of which may be sensitive to changes in ionic strength. Moreover, water with elevated EC may influence irrigation practices in agricultural contexts, potentially affecting crop health and soil quality due to salt accumulation.

In conclusion, the EC analysis indicates moderate salinity levels in the wetland water with notable spatial variability and a tendency toward higher values. The left-skewed and platykurtic nature of the data distribution suggests consistent inputs of dissolved salts without extreme fluctuations. Continued monitoring and careful management of land use and runoff sources are recommended to maintain the wetland system's ecological integrity and water quality.

iii. Total Dissolved Solids (TDS)

Total Dissolved Solids (TDS) measure the total concentration of dissolved organic and inorganic substances in water. These substances include minerals, salts, metals, cations, and anions naturally present or introduced through anthropogenic activities. TDS is a crucial water quality parameter because it affects water taste, suitability for drinking, agricultural use, and aquatic ecosystems. In this study, the TDS values in the water samples ranged from 26.70 mg/L to 150.60 mg/L, with an average value of 107.08 ± 13.32 mg/L. This mean TDS concentration suggests that the water contains moderate to relatively high dissolved solids. Although the values remain well below the World Health Organization's (WHO) recommended maximum limit of 1000 mg/L for drinking water, the elevated mean indicates substantial dissolved substances warranting further investigation.

The elevated TDS levels may be attributed to several environmental and anthropogenic sources. Soil erosion can contribute significantly to mineral content as surface soil particles rich in minerals dissolve and wash into water bodies. Agricultural runoff is another likely contributor, as fertilizers and pesticides applied to farmlands dissolve and increase the dissolved solids content in nearby streams and groundwater. Furthermore, natural processes such as the decomposition of organic matter release various dissolved compounds into the water. These combined factors contribute to the observed variability and average TDS concentration.

The observed standard deviation of 46.12 mg/L indicates a wide spread of TDS concentrations across the sampled sites. Such variability reflects spatial and temporal heterogeneity in the sources and processes affecting water quality. Areas closer to agricultural land or regions experiencing more intense rainfall may show elevated TDS due to increased runoff. In contrast, other areas may have lower values due to dilution or fewer inputs of dissolved solids. This variability emphasizes monitoring

multiple locations over time to capture a comprehensive picture of water quality dynamics.

Statistical analysis of the distribution of TDS values further enriches the understanding of the data. The skewness of -0.92 ± 0.64 suggests a left-skewed distribution. A negative skewness indicates that the tail on the lower end (i.e., lower TDS values) is longer or more pronounced than the upper end. This pattern implies that higher TDS concentrations are more common, which could point to consistent and persistent sources of dissolved solids in the water. These sources might include continuous agricultural runoff or steady soil leaching processes.

The kurtosis value of -0.90 ± 1.23 indicates a platykurtic distribution, meaning the TDS data distribution is relatively flat with lighter tails than a normal distribution. This suggests fewer extreme values or outliers, indicating that most TDS values cluster around the mean but are relatively evenly spread. A platykurtic distribution may result from a balance between natural fluctuations in dissolved solids and human activities, leading to a more homogeneous water quality profile without extreme contamination events.

Understanding these statistical characteristics is essential for water resource management and environmental protection. The moderate to high TDS levels, combined with their variability and distribution shape, can inform decisions regarding the suitability of the water for various uses. For instance, water with elevated TDS may affect irrigation efficiency, soil quality, and crop health. Similarly, high dissolved solids can impact aquatic life by altering water chemistry, potentially stressing sensitive species. In summary, the TDS analysis reveals that the water samples have moderate to relatively high levels of dissolved solids with significant variability across locations. The statistical distribution patterns suggest persistent sources of dissolved substances, likely linked to environmental factors such as soil erosion, agricultural activities, and organic matter decomposition. Continuous monitoring and targeted management interventions are recommended to mitigate potential impacts on water quality and ecosystem health.

iv. Salinity

Salinity is a key indicator of the concentration of dissolved salts in water, playing a vital role in defining the chemical characteristics and overall quality of aquatic ecosystems. It influences water chemistry, biological processes, and the suitability of water for various uses, including agriculture and habitat sustainability. Understanding salinity levels is particularly important in wetland environments, where fluctuating inputs from natural and anthropogenic sources can cause significant variations.

This study measured salinity values across the samples, ranging from 28.90 mg/L to 161.60 mg/L, with a mean concentration of 116.28 ± 14.31 mg/L. This relatively high average salinity suggests substantial inputs of dissolved salts, which may predominantly originate from agricultural runoff. Such runoff often carries inorganic fertilizers, including nitrates and phosphates, and other salts that accumulate in the surrounding soils and watercourses. These inputs can elevate salinity levels, affecting the wetland's ecological balance and potentially impacting the health of aquatic flora and fauna.

The standard deviation of 49.57 mg/L indicates considerable variability in salinity across the sampled sites. This wide range of salinity levels reflects the dynamic and heterogeneous nature of the wetland environment, where different areas may experience varying degrees of exposure to saline water. Factors such as rainfall patterns, soil types, land use, and proximity to pollution sources can all influence these variations, resulting in spatial and temporal fluctuations in salt concentration.

The skewness value of -0.95 ± 0.64 suggests that the salinity data are left-skewed, meaning that most salinity measurements are concentrated towards the higher end of the scale. This distribution pattern indicates frequent and consistent inputs of saline water or dissolved salts, possibly due to repeated agricultural activities or persistent natural processes such as mineral weathering or evaporation that concentrate salts in the wetland.

Furthermore, the kurtosis of -0.86 ± 1.23 indicates a platykurtic distribution. This means the salinity values are more broadly dispersed around the mean with fewer extremely high or low values than a normal distribution. Such a spread implies that while salinity fluctuates significantly, extreme spikes or drops in salt concentration

are relatively uncommon, possibly reflecting a balance between salt inputs and dilution or flushing mechanisms within the wetland. In summary, the salinity analysis reveals a wetland system characterized by moderately high and variable salt concentrations, with a tendency towards elevated salinity levels due to ongoing saline inputs. These findings underscore the importance of monitoring and managing salt sources to safeguard the wetland's ecological integrity and maintain its role as a valuable environmental resource.

v. Turbidity

Turbidity is a critical parameter in assessing water quality, as it measures the degree of cloudiness or haziness caused by suspended particles such as silt, clay, and organic matter. These suspended solids affect light penetration, which in turn influences aquatic plant photosynthesis and the overall health of the marine ecosystem. In the current analysis, turbidity values across the sampled wetland sites ranged from 108.20 to 194.60 Nephelometric Turbidity Units (NTU), with a mean turbidity of 160.38 ± 8.53 NTU (Table 4.29). This relatively high average turbidity indicates a substantial presence of particulate matter in the wetland water. Such elevated turbidity levels may result from various sources, including surface runoff carrying soil particles, riverbanks or adjacent land erosion, and the decomposition of organic materials within the wetland environment.

The standard deviation of 29.55 NTU reflects significant variability in turbidity measurements across the samples. This variation can be attributed to fluctuating environmental conditions such as changes in rainfall intensity, which may increase runoff and sediment transport, as well as human activities like agriculture or construction that disturb the soil and increase sediment load. Such variability is typical in wetland ecosystems, where natural processes and anthropogenic influences dynamically affect sediment suspension.

The skewness value of -0.51 ± 0.64 suggests a slight left-skewed distribution of turbidity data, meaning that most values tend to be on the higher end of the range. This indicates that suspended particulate matter is generally consistent, with fewer instances of low turbidity. Such a distribution aligns with expectations in wetlands where sediment input is relatively frequent.

Moreover, the kurtosis value of -1.06 ± 1.23 points to a platykurtic distribution, characterized by a flatter peak and broader spread of turbidity values compared to a normal distribution. This implies that the turbidity levels are more evenly dispersed, with fewer extreme highs or lows, suggesting a relatively balanced but variable sediment load in the water column over the sampling period. The descriptive statistical analysis of the key water quality parameters highlights the diverse concentrations and distribution patterns within the wetland samples examined. Table 4.42 summarises the statistical results for all parameters tested. The mean values serve as indicators of central tendency, with the standard errors providing a measure of precision for these estimates.

The pH values indicate slightly acidic conditions, which may be influenced by organic matter decomposition and agricultural inputs. Electrical Conductivity (EC) and Total Dissolved Solids (TDS) suggest moderate to high levels of salinity and dissolved minerals, reflecting significant salt concentrations in the wetland water. The relatively high salinity levels may be linked to agricultural runoff or natural mineral deposits.

Table 4.29: Descriptive statistics for the Water Quality parameters

Descriptive Statistics										
	N	Minimum	Maximum	Mean		Std. Deviation	Skewness		Kurtosis	
	Statistic	Statistic	Statistic	Statistic	Std. Error	Statistic	Statistic	Std. Error	Statistic	Std. Error
pH	12	5.50	6.51	6.260	0.093	0.32	-1.57	0.637	1.67	1.23
EC	12	0.04	0.23	0.159	0.019	0.06	-0.92	0.637	-0.86	1.23
TDS	12	26.70	150.6	107.0	13.31	46.12	-0.92	0.637	-0.90	1.23
Salinity	12	28.90	161.6	116.2	14.30	49.56	-0.94	0.637	-0.85	1.23
Turbidity	12	108.2	194.6	160.3	8.53	29.55	-0.50	0.637	-1.05	1.23

Turbidity measurements point to a considerable presence of suspended particles, consistent with soil erosion, runoff, and organic decay inputs. Across most parameters, the skewness values indicate left-skewed distributions, meaning that higher values tend to dominate. In contrast, the kurtosis values reveal platykurtic tendencies, suggesting broader distributions with fewer extreme outliers. These findings underscore the complex and dynamic nature of the wetland ecosystem. Water quality is shaped by natural processes, such as organic matter decomposition and mineral weathering, alongside anthropogenic influences, such as agriculture, land use changes, and runoff. Continuous monitoring and comprehensive management strategies are essential to maintain the ecological integrity of this critical habitat.

4.9.2 ANOVA analysis of water quality parameters

ANOVA on water quality parameters was done based on the season, sampling points, and sampling levels.

i. ANOVA for water quality parameters by season

The analysis of pH values across the dry and wet seasons reveals subtle but noteworthy seasonal variations in the acidity of the wetland water (Figure 4.24). During the dry season, the mean pH was recorded at 6.35 ± 0.11 , with observed values ranging from 5.81 to 6.51. This indicates a generally slightly acidic to near-neutral environment during the drier months. The standard deviation of 0.27 within the dry season suggests

moderate variability in pH values, implying that the water chemistry remains relatively stable during this period with only modest fluctuations.

In contrast, the wet season exhibited a slightly lower mean pH of 6.18 ± 0.15 , with pH values spanning a wider range from 5.50 to 6.51. This slight decrease in mean pH during the wet season may be attributed to increased inputs of acidic substances, possibly from organic matter decomposition and enhanced agricultural runoff due to rainfall. The higher standard deviation of 0.38 in the wet season reflects greater variability in pH values, indicating that the wet season water chemistry is more dynamic and influenced by variable environmental conditions such as precipitation intensity and land runoff. When considering both seasons combined, the overall mean pH was calculated as 6.26 ± 0.09 , reaffirming the wetland's slightly acidic nature. The combined data reflect a moderate variability in pH levels, underscoring the influence of seasonal changes on the chemical environment of the wetland ecosystem.

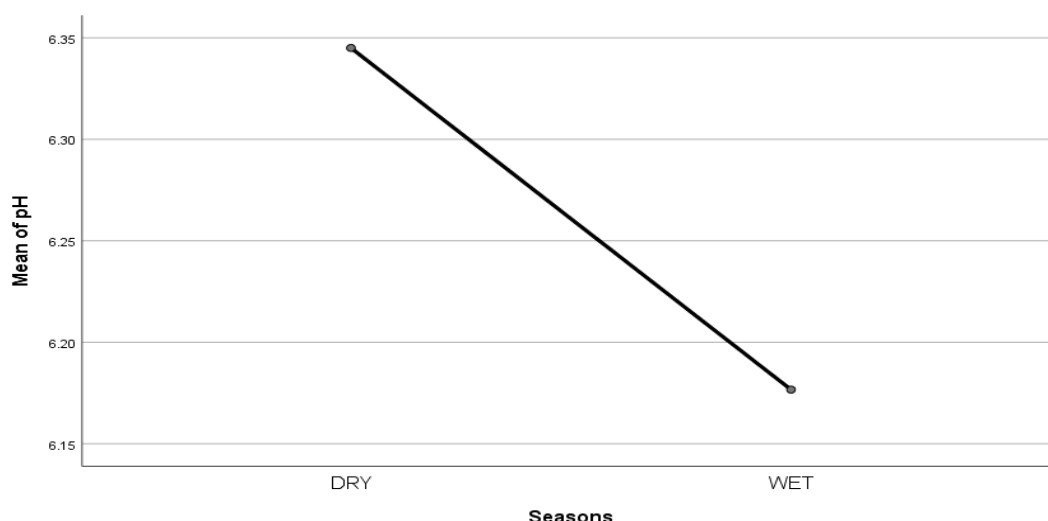


Figure 4.24: Seasonal pH mean of water samples

The analysis of electrical conductivity (EC) values across the dry and wet seasons reveals a consistent pattern in the salinity levels of the wetland water throughout the year. During the dry season, the mean EC was measured at 0.16 ± 0.03 mS/cm, with values ranging from 0.04 to 0.22 mS/cm. This range reflects moderate salinity levels, which may be influenced by limited dilution effects and ongoing mineral inputs during drier months.

Similarly, in the wet season, the mean EC remained at 0.16 ± 0.03 mS/cm, with a slightly broader range of 0.05 to 0.23 mS/cm. The similarity in mean EC values between the two seasons suggests that the wetland maintains relatively stable conductivity levels despite increased water volume and potential runoff during the wet season. However, the slight expansion in range during the wet season might reflect sporadic inputs of dissolved salts from surface runoff or agricultural activities.

The combined overall mean EC for both seasons is 0.16 ± 0.02 mS/cm, reinforcing the observation of moderate and stable salinity in the wetland water throughout the year (Figure 4.25). The standard deviation for EC remained consistent at 0.07 in both the dry and wet seasons, indicating a steady level of variability in dissolved salt concentrations regardless of seasonal changes. This consistency suggests that the wetland ecosystem experiences similar salinity dynamics year-round, potentially due to a balance between dilution during rains and salt accumulation during dry periods.

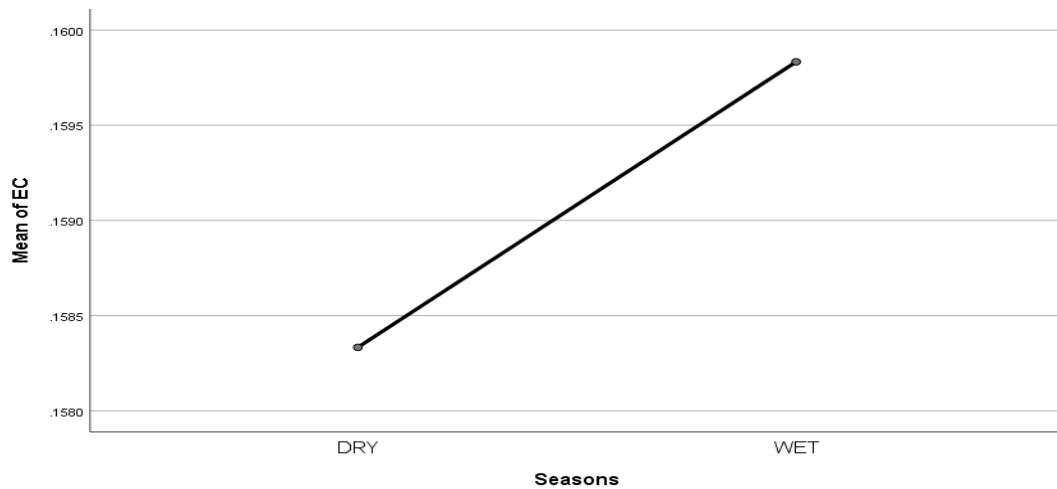


Figure 4.25: Seasonal EC mean of water samples

The analysis of Total Dissolved Solids (TDS) concentrations across the dry and wet seasons indicates relatively stable mean values throughout the year, with considerable variability among individual samples. During the dry season, the mean TDS was recorded at 106.97 ± 20.04 mg/L, with values ranging from 26.70 to 148.30 mg/L (Figure 4.26). This suggests that while the average concentration of dissolved solids remains moderately high, some sampling points exhibit significantly lower or higher

TDS levels, potentially reflecting localized influences such as soil mineral content or anthropogenic inputs.

Similarly, the wet season showed a mean TDS of 107.18 ± 19.46 mg/L, with a comparable range from 31.70 to 150.60 mg/L. The near-identical average values between the two seasons indicate that seasonal changes have minimal impact on the overall dissolved solids concentration in the wetland water. However, a broad range during the wet season may be attributed to increased surface runoff, which can introduce dilute water and concentrated dissolved substances depending on land use and rainfall intensity.

When both seasons are considered together, the total mean TDS is 107.08 ± 13.32 mg/L, reflecting a consistent presence of dissolved substances in the wetland environment. The overall standard deviation of approximately 46.12 mg/L highlights significant variability in TDS levels across all samples, underscoring the dynamic nature of the wetland's water chemistry. Such variability may arise from fluctuating environmental factors, including rainfall patterns, soil erosion rates, agricultural runoff, and organic matter decomposition, all of which can influence dissolved solid concentrations.

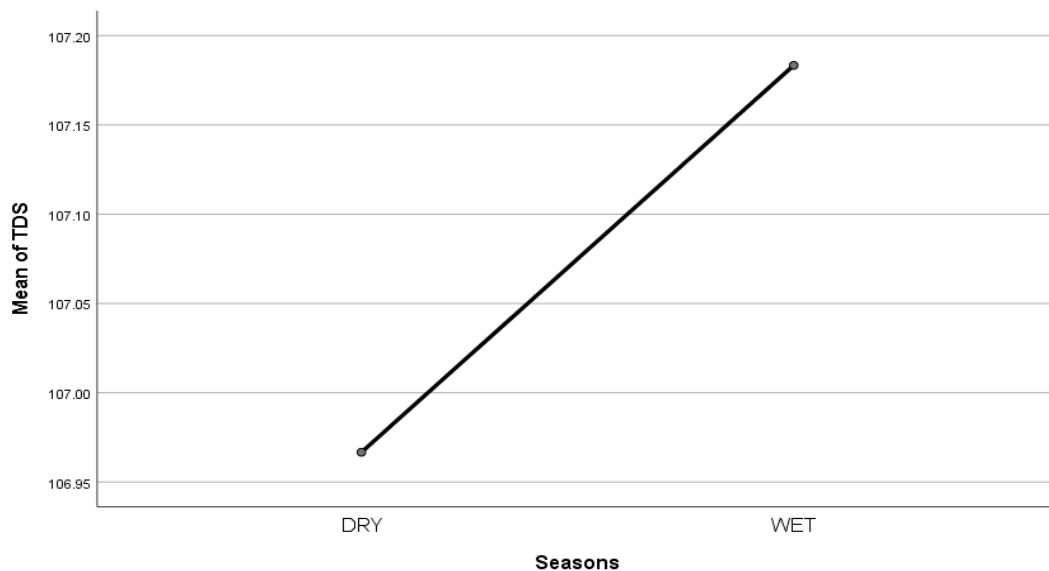


Figure 4.26: Seasonal TDS mean of water samples

The analysis of salinity levels in the wetland samples across the dry and wet seasons reveals a pattern of relatively high and variable salt concentrations throughout the year. During the dry season, the mean salinity was measured at 117.42 ± 22.01 mg/L, with

observed values ranging from 28.90 to 159.60 mg/L. This relatively elevated average suggests significant salt presence, potentially influenced by limited water dilution and the accumulation of salts in the environment during drier periods. In the wet season, the mean salinity was slightly lower at 115.15 ± 20.39 mg/L, with a broader range of values extending from 35.20 to 161.60 mg/L. The slight decrease in mean salinity during the wet season could be attributed to dilution effects from increased rainfall, which tends to reduce salt concentrations. However, the wider range indicates that intermittent inputs, such as runoff carrying salts from agricultural fields or soil erosion, contribute to fluctuating salinity levels during wetter months.

Considering both seasons together, the overall mean salinity stands at 116.28 ± 14.31 mg/L, reinforcing the notion of persistently moderate to high salinity within the wetland environment (Figure 4.27). The standard deviation of approximately 49.57 mg/L reflects considerable variability in salinity levels across all samples, suggesting dynamic and heterogeneous influences on salt concentrations. Such variability is likely driven by natural processes, including evaporation, precipitation, soil interactions, and anthropogenic factors such as fertilizer application and land use practices.

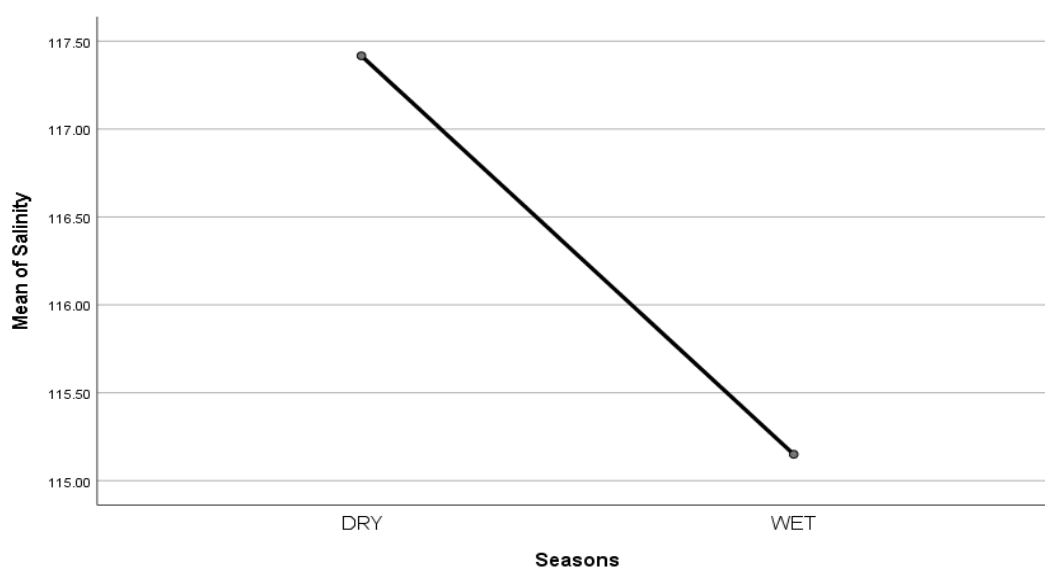


Figure 4.27: Seasonal Salinity mean of water samples

The turbidity measurements across the dry and wet seasons highlight notable differences in water clarity within the wetland ecosystem. During the dry season, turbidity values averaged 156.87 ± 9.73 NTU, from 122.20 to 192.20 NTU (Figure 4.28). These relatively high turbidity levels indicate a substantial presence of

suspended particles such as silt, clay, and organic matter. This may be attributed to soil erosion, limited water flow, or decomposition processes prevalent during drier conditions.

In the wet season, the average turbidity increased to 163.88 ± 14.86 NTU, with a broader range of 108.20 to 194.60 NTU. The elevated mean turbidity during this period likely reflects increased runoff and surface water flow, which can transport greater sediments and particulate matter into the wetland. The broader range of turbidity values in the wet season suggests fluctuating environmental conditions influenced by rainfall intensity, land use practices, and episodic disturbances.

When both seasons are considered together, the overall mean turbidity stands at 160.38 ± 8.53 NTU, signifying consistently high levels of suspended particles in the wetland waters throughout the year. The standard deviation of 29.55 highlights significant variability in turbidity across all samples, pointing to a dynamic system where particulate matter concentrations can vary widely due to natural seasonal cycles and human activities.

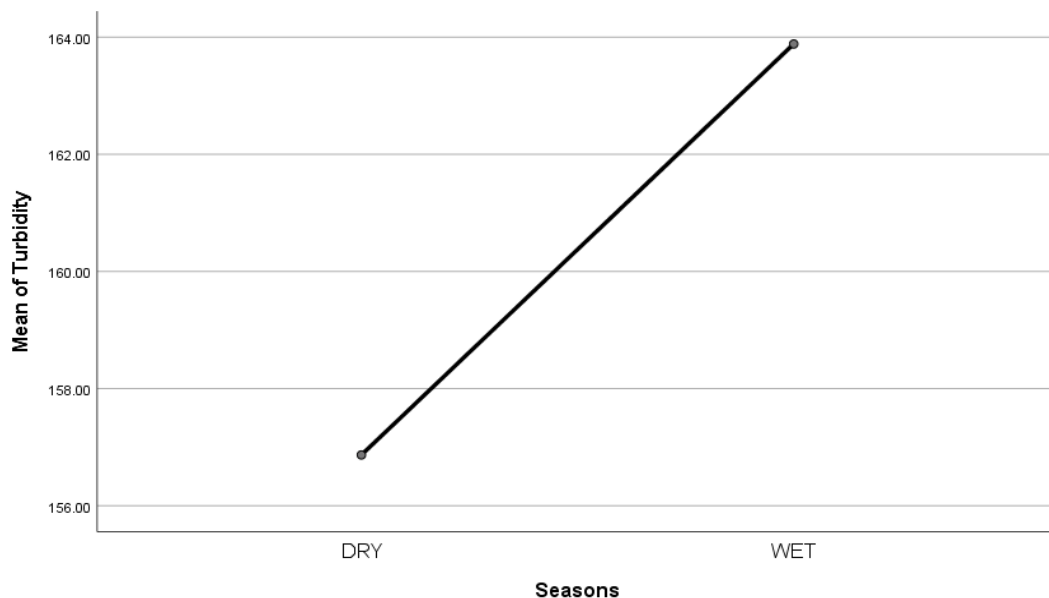


Figure 4.28: Seasonal Turbidity mean of water samples

The one-way Analysis of Variance (ANOVA) was conducted to determine whether there were statistically significant differences in selected water quality parameters between the dry and wet seasons. The results are presented in Table 4.30. The between-group sum of squares for pH was 0.085 with 1 degree of freedom (df), yielding a mean

square value of 0.085. The within-group sum of squares was 1.062 with 10 df, resulting in a mean square of 0.106. The computed F-value was 0.800, with a corresponding significance level (p-value) of 0.392. This indicates that the difference in pH levels between the dry and wet seasons is not statistically significant at the 0.05 level.

In the case of Electrical Conductivity (EC), the between-group sum of squares was 0.000 with one df, while the within-group sum of squares was 0.052 with 10 df. The respective mean squares were 0.000 and 0.005, producing an F-value of 0.001 and a p-value of 0.972. These results suggest that there is no significant seasonal variation in EC levels. Regarding Total Dissolved Solids (TDS), the analysis revealed a between-group sum of squares of 0.141 with one df and a within-group sum of squares of 23,402.362 with 10 df. The resulting mean square values were 0.141 and 2,340.236, respectively. The F-value was calculated at 0.000, and the p-value was 0.994. Thus, the seasonal differences in TDS concentrations are statistically insignificant. For salinity, the ANOVA showed a between-group sum of squares of 15.413 with one df and a within-group sum of squares of 27,008.383 with 10 df. The mean square values were 15.413 and 2,700.838, respectively, leading to an F-value of 0.006 and a p-value of 0.941. This outcome demonstrates no significant difference in salinity levels between the dry and wet seasons.

Finally, the ANOVA for turbidity yielded a between-group sum of squares of 147.701 with one df and a within-group sum of squares of 9,460.502 with 10 df. The corresponding mean squares were 147.701 and 946.050. The F-value was found to be 0.156, with a p-value of 0.701, again indicating no statistically significant difference in turbidity across the two seasons.

Table 4.30: ANOVA on water quality parameters by seasons

ANOVA						
		Sum of Squares	Df	Mean Square	F	Sig.
pH	Between Groups	0.085	1	0.085	0.800	0.392
	Within Groups	1.062	10	0.106		
	Total	1.147	11			
EC	Between Groups	0.000	1	0.000	0.001	0.972
	Within Groups	0.052	10	0.005		
	Total	0.052	11			
TDS	Between Groups	0.141	1	0.141	0.000	0.994
	Within Groups	23402.362	10	2340.236		
	Total	23402.503	11			
Salinity	Between Groups	15.413	1	15.413	0.006	0.941
	Within Groups	27008.383	10	2700.838		
	Total	27023.797	11			
Turbidity	Between Groups	147.701	1	147.701	0.156	0.701
	Within Groups	9460.502	10	946.050		
	Total	9608.202	11			

The one-way ANOVA results show no statistically significant seasonal differences in the analyzed water quality parameters pH, EC, TDS, salinity, or turbidity. This suggests that the wetland maintains relatively consistent water quality throughout the year, with natural and anthropogenic influences not producing significant seasonal shifts in these parameters.

ii. ANOVA for water quality parameters by sampling stations

The pH levels observed at the Maragua station demonstrate a mean value of 6.09 ± 0.16 , with recorded values ranging from 5.50 to 6.45. These values indicate slightly acidic to near-neutral conditions within this part of the wetland ecosystem (Figure 4.29). The relatively wide range and a standard deviation of 0.38 suggest notable variability in pH levels at Maragua. Such fluctuations may be attributed to environmental factors such as organic matter decomposition, variable runoff from

surrounding land, and potential inflows of acidic substances from nearby agricultural activities. These influences could introduce inconsistencies in water chemistry over time.

At the Mathioya station, the pH levels are comparatively higher and more stable, with a mean value of 6.43 ± 0.04 and a narrower range of 6.22 to 6.51. The standard deviation of 0.11 reflects limited variation, suggesting relatively uniform pH conditions at this location. The higher and more consistent pH values may indicate less exposure to acidic inputs or a more buffered aquatic environment, possibly due to differences in land use patterns, vegetation cover, or hydrological dynamics compared to the Maragua station.

When combining the data from both stations, the mean pH across the study area is 6.26 ± 0.09 , signifying that the wetland waters are generally slightly acidic, though close to neutral. The observed spatial differences in pH levels between the two stations underscore the influence of localized environmental conditions on water quality. This variation is essential for understanding ecosystem dynamics, as even slight shifts in pH can affect nutrient availability, microbial activity, and the overall health of aquatic organisms within the wetland system.

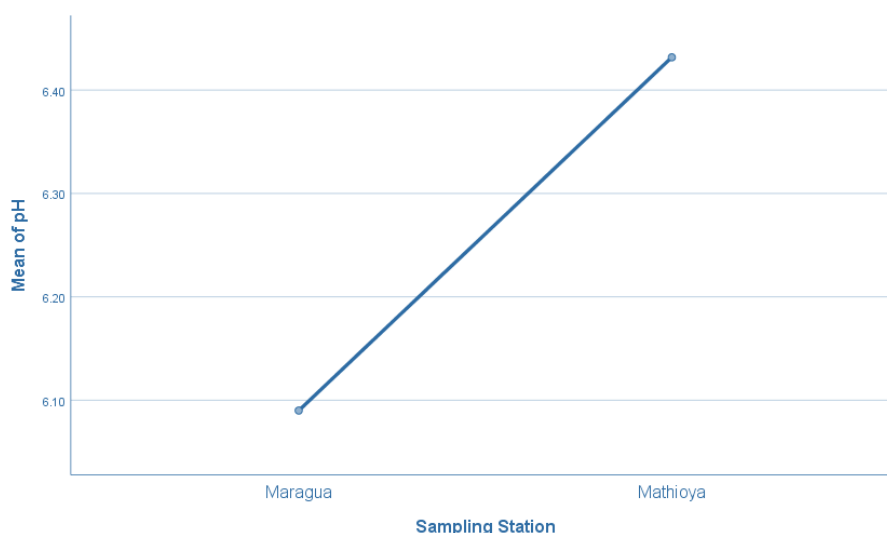


Figure 4.29: pH means for Maragua and Mathioya wetlands

The electrical conductivity (EC) levels, which indicate the ionic concentration and overall salinity of water, vary slightly between the two monitoring stations (Figure 4.30). At the Maragua station, the mean EC is 0.17 ± 0.02 mS/cm, with values ranging

from 0.10 to 0.23 mS/cm. This range suggests moderate salinity levels, potentially influenced by localized factors such as agricultural runoff, mineral dissolution, or dissolved salts from upstream sources. The standard deviation of 0.06 indicates moderate variability in EC values, suggesting relatively stable yet somewhat fluctuating ionic conditions at this site.

Conversely, the Mathioya station exhibits a slightly lower mean EC of 0.14 ± 0.03 mS/cm, with a broader range from 0.04 to 0.20 mS/cm. This lower average may reflect reduced input of dissolved salts or differing catchment characteristics, such as less anthropogenic influence or a more vegetated buffer zone. The standard deviation of 0.08 at Mathioya indicates a similar level of variability as at Maragua, reinforcing the idea that EC levels are influenced by dynamic and possibly seasonal factors across both stations. The overall mean EC combining data from both stations is 0.16 ± 0.02 mS/cm, reflecting a moderate level of electrical conductivity within the wetland system. This suggests that while there are minor spatial differences, the general ionic composition of the water remains within a consistent range. These findings highlight the role of local environmental conditions and catchment activities in shaping wetland waters' salinity and conductivity profiles.

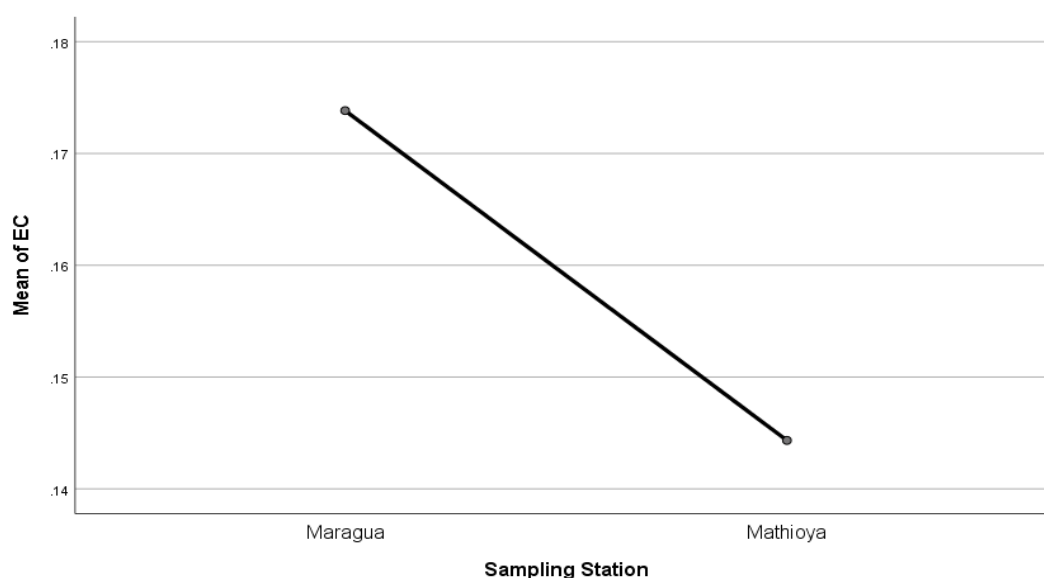


Figure 4.30: Mean of EC in Maragua and Mathioya wetlands

The Total Dissolved Solids (TDS) concentrations, which reflect the combined content of all inorganic and organic substances dissolved in water, show notable spatial variation between the two sampling stations (Figure 4.31). At the Maragua station, the mean TDS is 117.40 ± 16.67 mg/L, with observed values ranging from 64.90 to 150.60

mg/L. These concentrations suggest moderate levels of dissolved substances, potentially arising from upstream anthropogenic activities, natural mineral weathering, or leaching from surrounding soils. The standard deviation of 40.84 mg/L indicates a moderate variability in TDS levels at this station, which may be attributed to episodic inputs such as runoff events or fluctuations in groundwater interaction.

In contrast, the Mathioya station records a lower average TDS of 96.75 ± 21.44 mg/L, with a broader range of 26.70 to 133.90 mg/L. The reduced mean value may suggest lesser influence from anthropogenic sources or differing geological and hydrological conditions within the catchment area. However, the standard deviation of 52.51 mg/L at Mathioya is higher than at Maragua, indicating greater variability in TDS concentrations. This could reflect irregular influxes of dissolved materials, potentially due to spatial heterogeneity in land use, variable streamflow conditions, or periodic contamination events.

The overall mean TDS across both stations is 107.08 ± 13.32 mg/L, indicating that, on average, the water contains moderate dissolved solids. The variability in TDS across the two sites underscores the influence of both natural processes and human activities on the chemical composition of wetland water. These findings emphasize the importance of site-specific monitoring and the need to consider spatial variability when evaluating water quality across a broader ecosystem.

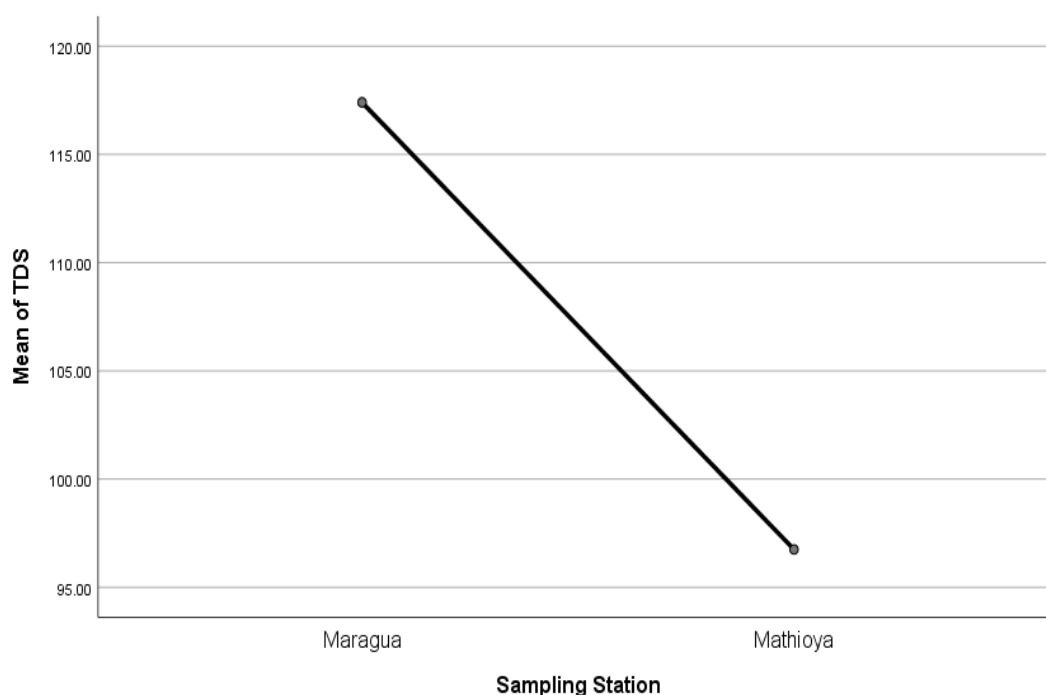


Figure 4.31: Mean of TDS in Maragua and Mathioya wetlands

The salinity levels—the concentration of dissolved salts in the water—exhibit apparent spatial differences between the two monitoring stations, reflecting varying environmental and anthropogenic influences (Figure 4.32). At the Maragua station, the mean salinity is measured at 126.03 ± 17.43 mg/L, with recorded values ranging from 71.70 to 161.60 mg/L. This relatively elevated salinity may be attributed to a combination of geological substrate, evapotranspiration effects, and possible agricultural runoff, which can introduce salts and minerals into the aquatic system. The standard deviation of 42.70 mg/L indicates a moderate to high level of variability in salinity, suggesting that the salinity levels at this station are subject to fluctuations likely driven by changes in land use or hydrological conditions.

In contrast, the Mathioya station reports a lower mean salinity of 106.53 ± 23.64 mg/L, with a broader range from 28.90 to 147.50 mg/L. This may point to reduced salt input or enhanced dilution from higher streamflow and precipitation in the region. Notably, the standard deviation is higher at this station—57.91 mg/L—which reflects considerable variability in salinity levels over time. Such variability may stem from episodic runoff, seasonal land-use practices, or varying soil salinity levels within the catchment.

The overall mean salinity across both stations is 116.28 ± 14.31 mg/L, a moderate concentration within acceptable limits for most freshwater ecosystems, although elevated values in some samples could warrant further investigation. The marked variability, especially at Mathioya, underscores the need for continuous monitoring to understand the drivers of salinity changes and their potential implications for aquatic biota, irrigation suitability, and wetland ecosystem health.

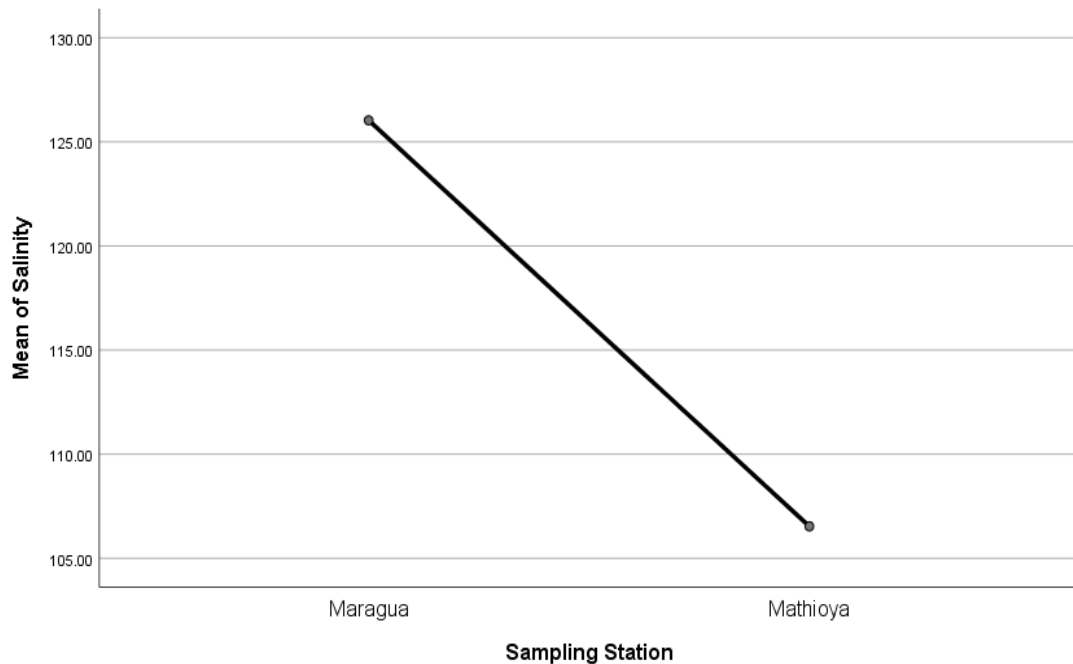


Figure 4.32: Mean of salinity in Maragua and Mathioya wetlands

Turbidity levels, which reflect the degree of water clarity and the presence of suspended particles such as silt, organic matter, and microorganisms, display notable variation between the two sampling stations (Figure 4.33). At the Maragua station, the mean turbidity is recorded at 159.23 ± 8.76 NTU, with values ranging from 128.30 to 183.80 NTU. These elevated turbidity values suggest a significant presence of suspended solids, potentially originating from surface runoff, soil erosion, or anthropogenic activities such as farming and deforestation in the catchment area. The standard deviation of 21.47 NTU at this station points to moderate variability, indicating that turbidity levels remain relatively stable but can fluctuate in response to episodic environmental events, such as storms or increased sediment input.

The Mathioya station shows a slightly higher mean turbidity of 161.52 ± 15.59 NTU, with a broader range of 108.20 to 194.60 NTU. The wider spread in values and a higher standard deviation of 38.18 NTU indicate greater variability in turbidity levels

at this site. This could be influenced by more dynamic hydrological processes, such as variable streamflow, more frequent disturbances in land cover, or differences in upstream land management practices. The elevated variability may also suggest episodic pollution events, particularly during or following rainfall, which can mobilize fine sediments and organic debris into the water column.

The overall mean turbidity across both stations stands at 160.38 ± 8.53 NTU, reflecting consistently high turbidity levels that could pose concerns for aquatic ecosystems, water filtration systems, and public health, especially if such values exceed regulatory thresholds. The notably higher variability at Mathioya underscores the need for site-specific strategies to manage sediment inputs and maintain water quality. Continuous monitoring and catchment-based interventions—such as buffer zones or erosion control, could help mitigate turbidity fluctuations and preserve aquatic habitat integrity.

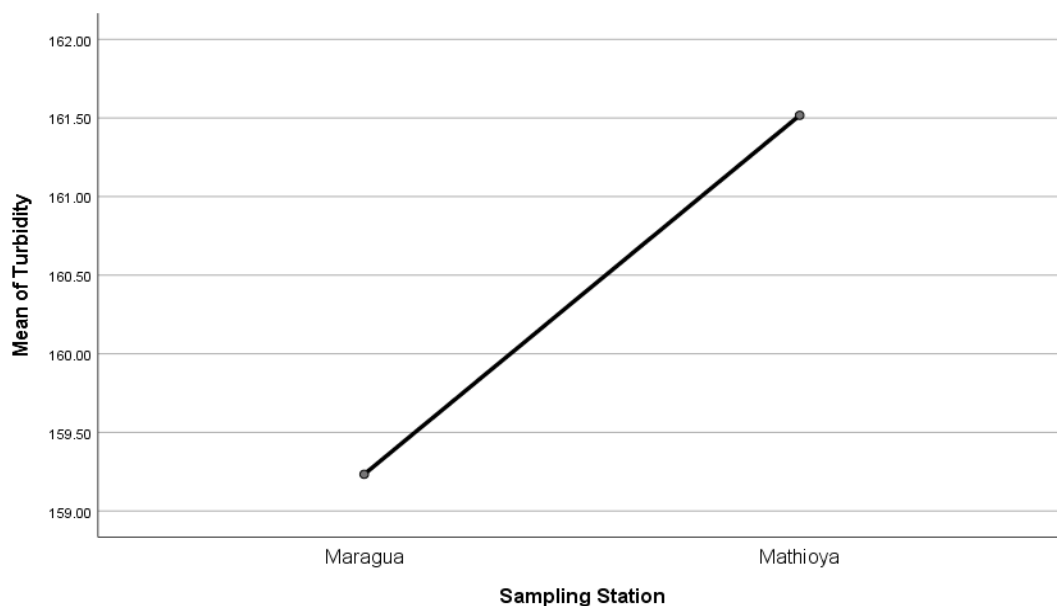


Figure 4.33: Mean of Turbidity in Maragua and Mathioya wetlands

A one-way analysis of variance (ANOVA) was conducted to examine whether there are statistically significant differences in selected water quality parameters—pH, Electrical Conductivity (EC), Total Dissolved Solids (TDS), salinity, and turbidity—between the Maragua and Mathioya stations. The results are as follows (Table 4.44): The ANOVA for pH reveals a between-group sum of squares of 0.350 with 1 degree

of freedom (df), resulting in a mean square of 0.350 as shown in Table 4.31. The within-group sum of squares is 0.797 with 10 df, yielding a mean square of 0.080. This results in an F-value of 4.395 and a p-value of 0.062. While this does not meet the conventional threshold of significance at $\alpha = 0.05$, the result is marginally non-significant, suggesting that there may be a potential difference in pH levels between the two stations. The result warrants further investigation, possibly with a larger sample size, as it approaches statistical significance and may have practical implications in water quality management.

For EC, the between-group sum of squares is 0.003 with one df, while the within-group sum is 0.050 with 10 df. The mean squares are 0.003 and 0.005, respectively, producing an F-value of 0.524 and a p-value of 0.486. These results indicate no statistically significant difference in electrical conductivity between Maragua and Mathioya stations. The observed similarities suggest a uniform ionic content distribution in the water at both sites.

The ANOVA for TDS reports a between-group sum of squares of 1279.267 and a within-group sum of squares of 22123.235, with mean squares of 1279.267 and 2212.324, respectively. The resulting F-value is 0.578, and the p-value is 0.465, indicating no significant difference in TDS levels between the two stations. This suggests that the concentration of dissolved inorganic and organic substances in the water remains relatively consistent across both locations.

The results for salinity show a between-group sum of squares of 1140.750 and a within-group sum of squares of 25883.047, with mean squares of 1140.750 and 2588.305, respectively. The F-value of 0.441 and p-value of 0.522 suggest that salinity levels do not significantly differ between the Maragua and Mathioya stations. This finding indicates a stable salt concentration across the catchment, possibly due to similar geological and hydrological conditions.

Finally, the ANOVA for turbidity reports a between-group sum of squares of only 15.641, while the within-group sum of squares is 9592.562. The corresponding mean squares are 15.641 and 959.256, yielding a very low F-value of 0.016 and a p-value of 0.901. These results confirm no statistically significant difference in turbidity

between the two sampling stations, suggesting a consistent load of suspended particles and sediments, potentially influenced by similar land-use practices upstream.

Except for pH, which shows a marginally non-significant difference, all other parameters—including EC, TDS, salinity, and turbidity—exhibit no statistically significant differences between the Maragua and Mathioya stations. These findings imply broadly comparable water quality conditions across both sites, reflecting similar environmental influences, hydrological connectivity, or shared anthropogenic impacts in the region. However, the near-significance of pH warrants further monitoring and could serve as an early signal of subtle changes in water chemistry between the stations.

Table 4.31: ANOVA on water quality parameters by stations

ANOVA by sampling stations						
		Sum of Squares	Df	Mean Square	F	Sig.
pH	Between Groups	0.350	1	0.350	4.395	0.062
	Within Groups	0.797	10	0.080		
	Total	1.147	11			
EC	Between Groups	0.003	1	0.003	0.524	0.486
	Within Groups	0.050	10	0.005		
	Total	0.052	11			
TDS	Between Groups	1279.267	1	1279.267	0.578	0.465
	Within Groups	22123.235	10	2212.324		
	Total	23402.503	11			
Salinity	Between Groups	1140.750	1	1140.750	0.441	0.522
	Within Groups	25883.047	10	2588.305		
	Total	27023.797	11			
Turbidity	Between Groups	15.641	1	15.641	0.016	0.901
	Within Groups	9592.562	10	959.256		
	Total	9608.202	11			

iii. **ANOVA Analysis of water quality parameters by sampling levels**

The pH levels across the three sampling levels—Down-Stream, Mid-Stream, and Up-Stream—show a distinct variation, suggesting spatial differences in water chemistry

along the river system (Figure 4.34). The mean pH is the highest at the Downstream section at 6.47 ± 0.03 , with values ranging narrowly from 6.40 to 6.51. This indicates slightly more alkaline conditions and a relatively stable pH environment. The low variability in this section may be attributed to the natural mixing and dilution of water as it flows downstream, potentially receiving cleaner tributaries or experiencing reduced pollution inputs compared to upstream areas.

In contrast, the Mid-Stream section exhibits a moderate mean pH of 6.31 ± 0.10 , with a broader range of 6.01 to 6.45. The increased variability here suggests that the water quality in the middle section of the river may be influenced by fluctuating environmental conditions or localized inputs such as agricultural runoff or semi-treated wastewater. This part of the river may serve as a transition zone where upstream and downstream influences converge, leading to a broader range of pH values.

The Up-Stream section shows the lowest mean pH at 6.00 ± 0.22 , ranging from 5.50 to 6.48. The higher variability indicates less stable pH conditions, possibly due to localized human activities, industrial discharge, or natural factors such as leaching from acidic soils or high levels of organic matter. These more acidic conditions upstream could pose potential stress to aquatic life and signal the need for closer monitoring and possible remediation.

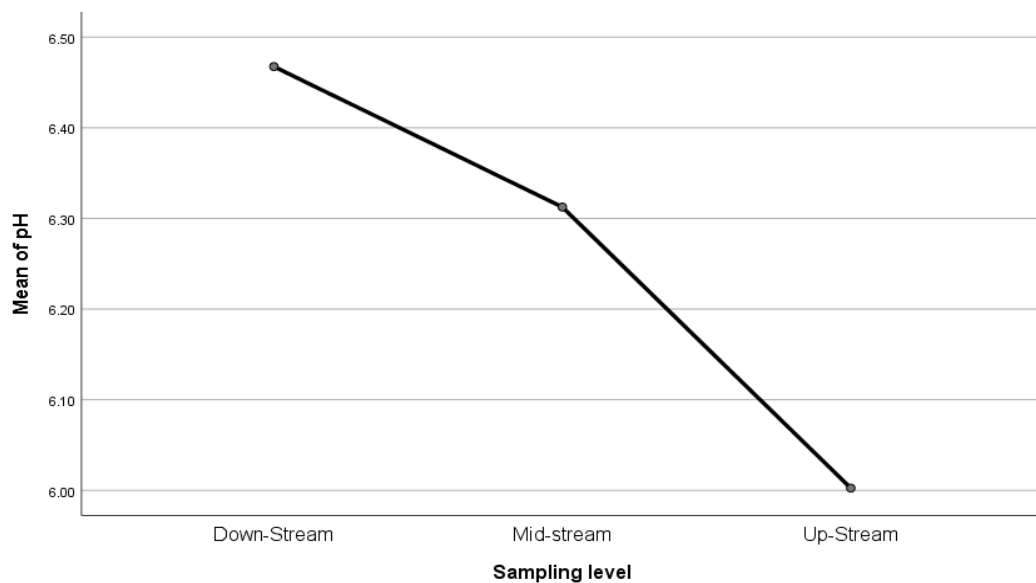


Figure 4.34: Mean of pH by sampling level

Electrical Conductivity (EC) levels demonstrate noticeable variation across the different sampling levels—Down-Stream, Mid-Stream, and Up-Stream highlighting differences in ionic concentration and overall water quality along the river (Figure 4.35). At the Downstream level, EC values are relatively high and stable, with a mean of 0.20 ± 0.00 mS/cm and a narrow range between 0.19 and 0.20 mS/cm. This suggests consistent ionic content, possibly due to cumulative inputs of dissolved solids and minerals from upstream and adjacent sources as the river flows downstream.

The Mid-Stream section has a slightly higher average EC at 0.21 ± 0.01 mS/cm and a broader range from 0.19 to 0.23 mS/cm. The slight increase in mean EC and the observed variability could indicate localized additions of dissolved salts, such as those from agricultural runoff, wastewater discharges, or natural mineral deposits. This segment likely experiences dynamic water quality changes influenced by human activities and natural processes, contributing to its variable conductivity profile.

Conversely, the Upstream area records a much lower mean EC of 0.07 ± 0.02 mS/cm, with values ranging from 0.04 to 0.10 mS/cm. This low conductivity indicates fewer dissolved ions, which may reflect minimal anthropogenic influence, cleaner water sources, or less mineralized geology. The broader range of EC values in this section suggests that while the area is generally low in conductivity, occasional events or conditions might introduce fluctuations in ion concentration.

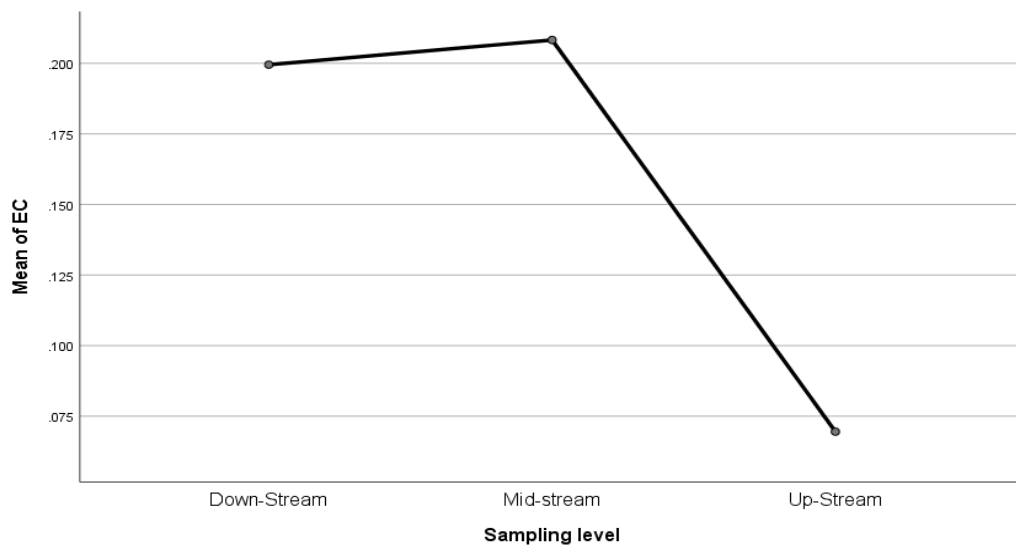


Figure 4.35: Mean of EC by sampling level

Total Dissolved Solids (TDS) levels exhibit marked differences across the various sampling levels, reflecting significant spatial variation in the concentration of

dissolved inorganic and organic substances in the water (Figure 4.36). At the Downstream level, the mean TDS is 135.43 ± 1.46 mg/L, with values ranging from 132.60 to 139.30 mg/L. This narrow range and relatively high mean suggest stable but elevated levels of dissolved solids, likely due to cumulative inputs from upstream sources, urban runoff, agricultural discharge, or natural mineral leaching as the river moves downstream.

In the Mid-stream section, the TDS mean is slightly higher at 138.63 ± 6.60 mg/L, with a broader range of 122.70 to 150.60 mg/L. The increased variability in this section may indicate intermittent contributions from tributaries, settlements, or land use activities that introduce fluctuating levels of dissolved materials. Seasonal patterns, land management practices, or localized pollution sources could drive these changes.

Conversely, the Upstream area shows significantly lower TDS levels, with a mean of 47.18 ± 10.43 mg/L and a range between 26.70 and 65.40 mg/L. These lower values reflect a comparatively pristine section of the river with minimal anthropogenic disturbance and limited mineral content. This zone's higher standard deviation and wider range may be attributed to natural factors such as geological variation or episodic inputs from rainfall or surface runoff.

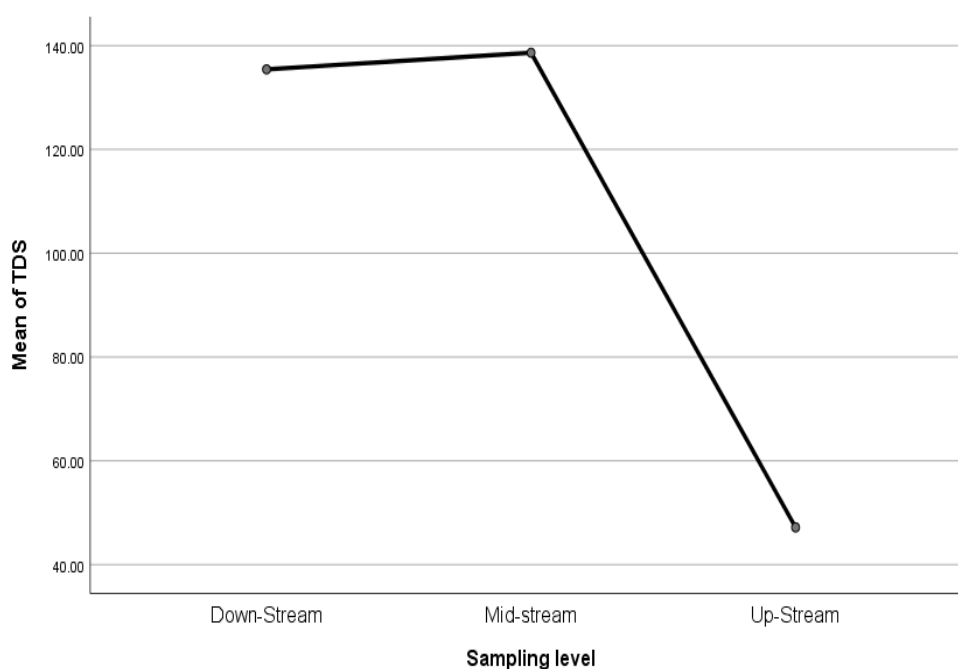


Figure 4.36: Mean of TDS by sampling level

Salinity levels showed significant spatial variation across the different sampling points along the river, reflecting the influence of both natural processes and anthropogenic activities. In the Downstream section, the mean salinity was recorded at 146.05 ± 1.81 mg/L, with values ranging from 141.40 to 150.20 mg/L (Figure 4.37). This indicates relatively stable and elevated salinity concentrations in this zone, likely due to dissolved salts accumulated from upstream sources.

In the Mid-stream region, the mean salinity was slightly higher at 150.93 ± 6.15 mg/L, with a broader range of 135.00 to 161.60 mg/L. The increased variability here may be attributed to agricultural activities commonly practiced in this zone. Using inorganic fertilizers in farms adjacent to the river contributes to salinity spikes, as these fertilizers are carried into the water system through surface runoff during rainfall or irrigation events. The presence of high concentrations of dissolved salts from fertilizers results in elevated salinity levels detectable in this segment of the river.

The Upstream sampling area, in contrast, recorded a significantly lower mean salinity of 51.88 ± 11.52 mg/L, with values ranging from 28.90 to 71.70 mg/L. This can be explained by the minimal agricultural interference in the upstream wetlands, where the landscape is generally more forested and less disturbed by human activity. As a result, fewer salts and chemical residues enter the water through runoff, keeping salinity levels relatively low.

Turbidity levels displayed noticeable variability across the three sampling locations, indicating differences in the concentration of suspended particles along the river course. At the Downstream section, turbidity averaged 170.50 ± 15.40 NTU, with observed values ranging from 128.30 to 194.60 NTU. This relatively high mean and wide range suggest downstream areas receive significant sediment loads, likely due to accumulated runoff and disturbed riverbanks from upstream activities.

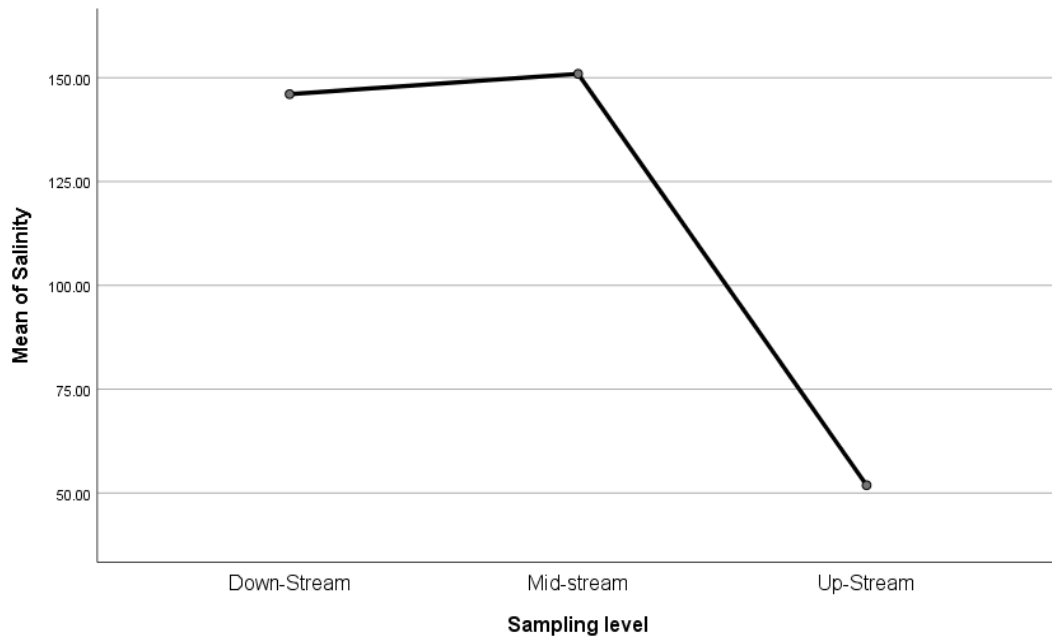


Figure 4.37: Mean of salinity by sampling level

In the Mid-stream region, turbidity levels were slightly higher, with a mean of 173.53 ± 8.13 NTU and a narrower range between 158.40 and 190.90 NTU (Figure 4.38). The reduced variation in this segment could be attributed to more consistent levels of erosion or land disturbance, possibly from ongoing agricultural or semi-urban activities. This consistency, however, does not imply improved water clarity. It points to a steady supply of suspended solids, likely from surface runoff and localized human activities.

The Upstream section exhibited lower turbidity levels, with a mean of 137.10 ± 15.00 NTU and a range from 108.20 to 177.50 NTU. These lower values may reflect the more pristine condition of the upstream environment, where forest cover, reduced human activity, and more stable soils minimize sediment input into the river. Nonetheless, occasional peaks within the upstream range suggest that some episodic disturbances, such as rainfall events or land clearing, may still impact turbidity.

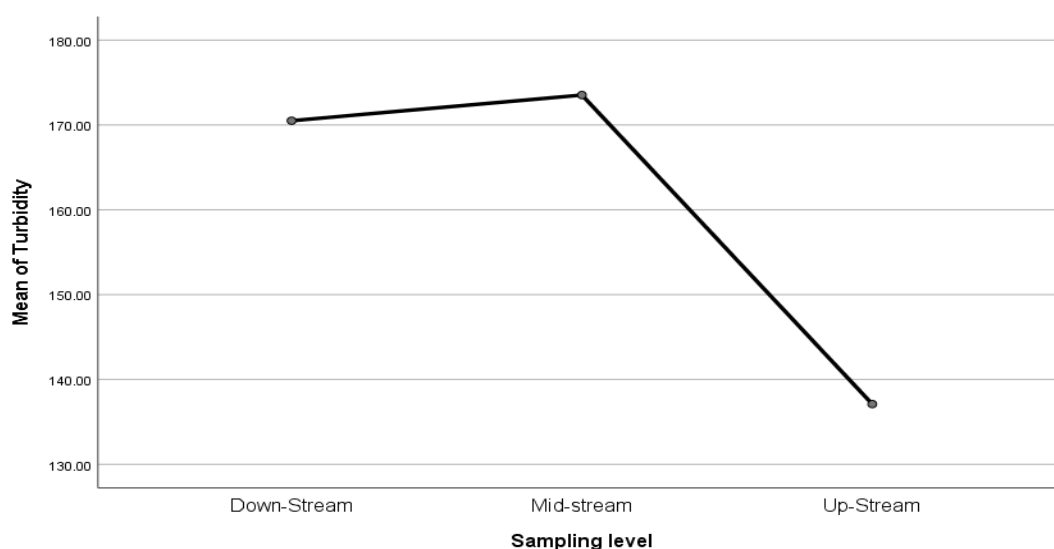


Figure 4.38: Mean of Turbidity by sampling level

Table 4.32 presents ANOVA results of water quality parameters by sampling levels: downstream, midstream, and upstream. The analysis of variance (ANOVA) for pH levels across the three sampling levels downstream, midstream, and upstream revealed a between-groups sum of squares of 0.448 with 2 degrees of freedom (df), yielding a mean square of 0.224. The within-groups sum of squares was 0.699 with nine df, resulting in a mean square of 0.078. The computed F-value was 2.889, and the corresponding significance level (p-value) was 0.107. This means that although there was observable variation in pH values across the different river sections, the differences were not statistically significant at the 0.05 level. The highest mean pH was recorded downstream (6.47), followed by midstream (6.31), and the lowest was upstream (6.00). The lack of statistical significance implies that these differences may be due to natural variability or measurement error rather than distinct environmental or anthropogenic influences. Nevertheless, the gradual increase in pH moving downstream could be associated with the cumulative effects of human activities or natural buffering processes along the river.

The ANOVA results for Electrical Conductivity (EC) showed significant differences between the sampling levels. The between-groups sum of squares was 0.048 with two df, while the within-groups sum was only 0.004 with nine df. These values resulted in mean squares of 0.024 and 0.000, respectively. The F-value was exceptionally high at 52.546, with a p-value of less than 0.001, indicating a significant difference in EC levels across the upstream, midstream, and downstream locations. Specifically, the

midstream had the highest mean EC (0.21 mS/cm), followed closely by downstream (0.20 mS/cm), while upstream exhibited significantly lower EC levels (0.07 mS/cm). This trend suggests a strong influence of anthropogenic factors, such as runoff from agricultural or urban areas, contributing to increased ion concentrations midstream and downstream. The highly significant F-value underscores the reliability of this observation, suggesting that conductivity is a sensitive indicator of environmental disturbances and solute loading along the river's flow path.

For Total Dissolved Solids (TDS), the ANOVA revealed a between-groups sum of squares of 21,548.540 with two df and a within-groups sum of squares of 1,853.963 with nine df. The resulting mean squares were 10,774.270 and 205.996, respectively. The F-value of 52.303, coupled with a p-value less than 0.001, indicates a statistically significant difference in TDS levels among the three sampling locations. The downstream and midstream regions recorded much higher TDS levels (135.43 mg/L and 138.63 mg/L, respectively), while upstream exhibited substantially lower values (47.18 mg/L). These differences are likely due to dissolved inorganic and organic matter introduced into the river through agricultural runoff, domestic wastewater, and possibly minor industrial discharges downstream and midstream. The sharp contrast between the upstream and the other sections highlights the relatively unpolluted nature of the upstream environment. It underscores the influence of land use on water quality as water flows downstream.

The ANOVA results for salinity levels also showed significant differences across sampling locations. The between-groups sum of squares was 24,938.132 with two df, and the within-groups sum was 2,085.665 with nine df. The resulting mean squares were 12,469.066 and 231.741, respectively. An F-value of 53.806 with a p-value less than 0.001 confirmed a significant salinity difference among the upstream, midstream, and downstream areas. The midstream had the highest mean salinity (150.93 mg/L), closely followed by downstream (146.05 mg/L), while upstream recorded the lowest (51.88 mg/L). These results are consistent with known patterns of nutrient enrichment in water bodies exposed to agricultural practices. The elevated salinity in the midstream and downstream locations can be attributed to fertilizer use and farm surface runoff, which introduces salts into the river system. These residues accumulate as the water flows downstream, leading to elevated salinity levels. In contrast, the

upstream region is less affected by such anthropogenic pressures, resulting in significantly lower salinity. The statistically significant F-value suggests that salinity is another sensitive parameter to track agricultural and land use impacts on riverine systems.

For turbidity, the ANOVA yielded a between-groups sum of squares of 3,268.655 with two df and a within-groups sum of squares of 6,339.548 with nine df. The respective mean squares were 1,634.328 and 704.394, which produced an F-value of 2.320. The p-value of 0.154 indicates that the differences in turbidity across sampling levels were not statistically significant. Although the midstream (173.53 NTU) and downstream (170.50 NTU) areas recorded higher mean turbidity levels compared to upstream (137.10 NTU), these differences were not sufficient to conclude that turbidity varies systematically across locations. This could be due to the naturally dynamic nature of turbidity, which may be influenced by temporal events such as rainfall, sediment disturbances, or vegetation cover rather than consistent land-use patterns. The relatively high within-group variability may have also diluted the statistical power to detect significant differences. Still, the elevated turbidity values downstream and midstream may indicate moderate to high suspended particle loads, likely from erosion and runoff, even if the differences were not statistically robust.

Table 4.32: ANOVA on water quality parameters by sampling levels

ANOVA by sampling levels						
		Sum of Squares	Df	Mean Square	F	Sig.
pH	Between Groups	0.448	2	0.224	2.889	0.107
	Within Groups	0.699	9	0.078		
	Total	1.147	11			
EC	Between Groups	0.048	2	0.024	52.546	0.000
	Within Groups	0.004	9	0.000		
	Total	0.052	11			
TDS	Between Groups	21548.540	2	10774.270	52.303	0.000
	Within Groups	1853.963	9	205.996		
	Total	23402.503	11			
Salinity	Between Groups	24938.132	2	12469.066	53.806	0.000
	Within Groups	2085.665	9	231.741		
	Total	27023.797	11			
Turbidity	Between Groups	3268.655	2	1634.328	2.320	0.154
	Within Groups	6339.548	9	704.394		
	Total	9608.203	11			

The post-hoc test revealed a significant difference in pH levels between Down-Stream and Up-Stream, with a mean difference of 0.465 ($p = .043$) (Table 4.33). This suggests that the water in the downstream area is slightly more alkaline compared to the upstream. The increased pH downstream may be due to domestic or agricultural runoff inputs that contain alkaline substances such as detergents or lime-based fertilizers. However, no statistically significant differences in pH were observed between Down-Stream and Mid-Stream or between Mid-Stream and Up-Stream. This implies that pH variations along the river are relatively minor and only become statistically distinct when comparing the most environmentally contrasting sections upstream (relatively pristine) and downstream (more exposed to human activity).

Highly significant differences in EC were observed between Down-Stream and Up-Stream (mean difference = 0.130, $p < .001$) and Mid-Stream and Up-Stream (mean

difference = 0.139, $p < .001$). These results indicate that midstream and downstream areas have higher ion concentrations than upstream, reflecting greater exposure to anthropogenic pollution, such as fertilizer runoff, sewage discharge, or urban waste. No significant difference in EC was detected between Down-Stream and Mid-Stream. This suggests that most ion-related inputs have already occurred when water flows past the midstream section, and the conductivity levels remain relatively consistent downstream.

Similar to EC, the post-hoc test showed statistically significant differences in TDS levels between Down-Stream and Up-Stream (mean difference = 88.250, $p < .001$) and Mid-Stream and Up-Stream (mean difference = 91.450, $p < .001$). This confirms that downstream and midstream waters contain considerably more dissolved solids, including minerals, organic matter, and pollutants, than upstream. The source of these solids could be linked to soil erosion, leaching of fertilizers, wastewater discharge, and surface runoff. However, no significant difference between Down-Stream and Mid-Stream indicated a plateauing effect in TDS concentrations beyond the midstream section.

The post-hoc results indicated significant differences in salinity between Down-Stream and Up-Stream (mean difference = 94.175, $p < .001$) and Mid-Stream and Up-Stream (mean difference = 99.050, $p < .001$). These results are consistent with the EC and TDS findings and suggest that salts and other soluble compounds accumulate as the river progresses downstream. The elevated salinity levels midstream and downstream may reflect agricultural and domestic activities, where irrigation return flows, fertilizers, and household effluents increase the ionic concentration. Again, no significant difference in salinity was found between the Downstream and Midstream, implying that the bulk of salinity-altering inputs occur before or within the midstream section.

No statistically significant differences were found between the turbidity sampling levels (Down-Stream, Mid-Stream, and Up-Stream). Although descriptive statistics suggested variability notably higher values midstream and downstream, this variation was not strong enough to achieve statistical significance. This could be due to high within-group variability, which reduces the statistical power to detect differences. Alternatively, turbidity may fluctuate based on short-term events (e.g., rainfall,

sediment resuspension) rather than long-term land use or pollution trends. This result suggests that while turbidity levels may visually appear to vary, these changes are not consistently different across the spatial gradient of the river in a statistically meaningful way.

Table 4.33: LSD on water quality parameters at various sampling levels

Multiple Comparisons							
LSD							
Dependent Variable	(I) Sampling level	(J) Sampling level	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
						Lower Bound	Upper Bound
Ph	Down-Stream	Mid-stream	0.15500	0.19701	0.452	-0.2907	0.6007
		Up-Stream	0.46500*	0.19701	0.043	0.0193	0.9107
	Mid-stream	Down-Stream	-0.15500	0.19701	0.452	-0.6007	0.2907
		Up-Stream	0.31000	0.19701	0.150	-0.1357	0.7557
	Up-Stream	Down-Stream	-0.46500*	0.19701	0.043	-0.9107	-0.0193
		Mid-stream	-0.31000	0.19701	0.150	-0.7557	0.1357
EC	Down-Stream	Mid-stream	-0.00875	0.01516	0.578	-0.0430	0.0255
		Up-Stream	0.13000*	0.01516	0.000	0.0957	0.1643
	Mid-stream	Down-Stream	0.00875	0.01516	0.578	-0.0255	0.0430
		Up-Stream	0.13875*	0.01516	0.000	0.1045	0.1730
	Up-Stream	Down-Stream	-0.13000*	0.01516	0.000	-0.1643	-0.0957
		Mid-stream	-0.13875*	0.01516	0.000	-.1730	-0.1045
TDS	Down-Stream	Mid-stream	-3.20000	10.14879	0.760	-26.1582	19.7582
		Up-Stream	88.25000*	10.14879	0.000	65.2918	111.2082

	Mid-stream	Down-Stream	3.20000	10.14879	0.760	-19.7582	26.1582
		Up-Stream	91.45000*	10.14879	0.000	68.4918	114.4082
	Up-Stream	Down-Stream	-88.25000*	10.14879	0.000	-111.2082	-65.2918
		Mid-stream	-91.45000*	10.14879	0.000	-114.4082	-68.4918
Salinity	Down-Stream	Mid-stream	-4.87500	10.76431	0.661	-29.2256	19.4756
		Up-Stream	94.17500*	10.76431	0.000	69.8244	118.5256
	Mid-stream	Down-Stream	4.87500	10.76431	0.661	-19.4756	29.2256
		Up-Stream	99.05000*	10.76431	0.000	74.6994	123.4006
	Up-Stream	Down-Stream	-94.17500*	10.76431	0.000	-118.5256	-69.8244
		Mid-stream	-99.05000*	10.76431	0.000	-123.4006	-74.6994
Turbidity	Down-Stream	Mid-stream	-3.02500	18.76691	0.876	-45.4787	39.4287
		Up-Stream	33.40000	18.76691	0.109	-9.0537	75.8537
	Mid-stream	Down-Stream	3.02500	18.76691	0.876	-39.4287	45.4787
		Up-Stream	36.42500	18.76691	0.084	-6.0287	78.8787
	Up-Stream	Down-Stream	-33.40000	18.76691	0.109	-75.8537	9.0537
		Mid-stream	-36.42500	18.76691	0.084	-78.8787	6.0287
*. The mean difference is significant at the 0.05 level.							

The descriptive statistics and ANOVA results for pH, electrical conductivity (EC), total dissolved solids (TDS), salinity, and turbidity demonstrate that significant spatial variations exist in EC, TDS, and salinity levels across the three sampling locations:

Down-Stream, Mid-Stream, and Up-Stream. pH levels exhibited minor variation, which was not statistically significant, while turbidity showed no significant differences among the sampling points. Post-hoc analysis provided further insight, revealing that the most notable differences in EC, TDS, and salinity occur between Up-Stream and the other two locations (Mid-Stream and Down-Stream). These differences likely reflect increased anthropogenic influence—particularly agricultural and domestic activities—affecting water chemistry as the stream flows downstream.

These findings indicate that water quality deteriorates progressively from upstream to downstream, with upstream areas maintaining comparatively better quality across several key parameters. This spatial pattern underscores the impact of land use and runoff on aquatic ecosystems, emphasizing the need for targeted water management strategies in midstream and downstream zones.

4.9.3 Correlation analysis for water quality parameters

The initial correlation analysis explored the relationships among key water quality parameters: pH, Electrical Conductivity (EC), Total Dissolved Solids (TDS), Salinity, and Turbidity. This analysis utilized Pearson's correlation coefficient, a statistical measure that assesses the strength and direction of the linear relationship between two continuous variables. The results, including correlation coefficients (r^2) and their corresponding significance levels (p-values), are summarized in Table 4.34.

Starting with pH, the analysis revealed generally weak and statistically non-significant correlations with the other water quality parameters. The strongest correlation involving pH was observed with salinity ($r^2 = 0.371$), but this association was not statistically significant ($p = 0.235$). This suggests that pH varies somewhat independently of the other measured parameters in this dataset, indicating that changes in pH are not strongly linked to variations in EC, TDS, salinity, or turbidity within the sampled locations.

In contrast, the variables EC, TDS, and salinity exhibited extremely high positive correlations. The correlation coefficients between each pair of these variables were exceptionally close to 1 ($r^2 = 0.999$), all significant at the 0.01 level. This near-perfect linear relationship indicates that these parameters are closely interconnected, likely reflecting their common dependence on dissolved ionic substances in the water. As

EC measures the water's ability to conduct electrical current, which is directly influenced by the concentration of dissolved salts and solids, it is expected to correlate strongly with TDS and salinity, both of which quantify the amount of dissolved materials in the water.

Turbidity, which measures the cloudiness or suspended particle content in water, showed moderate positive correlations with EC ($r^2 = 0.655$, $p = 0.021$), TDS ($r^2 = 0.628$, $p = 0.029$), and salinity ($r^2 = 0.661$, $p = 0.019$). All these correlations were statistically significant at the 0.05 level. This indicates that as the levels of dissolved solids and salts increase in the water, there tends to be a corresponding increase in turbidity. This relationship may suggest that areas with higher mineral and salt content also experience greater particulate matter or sediment presence, possibly due to runoff or other environmental factors affecting water clarity.

Table 4.34: Correlation analysis of water quality parameters

		Ph	EC	TDS	Salinity	Turbi dity
pH	Pearson Correlation	1	0.358	0.368	0.371	0.017
	Sig. (2-tailed)		0.253	0.239	0.235	0.958
	N	12	12	12	12	12
EC	Pearson Correlation	0.358	1	0.999**	0.999**	0.655*
	Sig. (2-tailed)	0.253		0.000	0.000	0.021
	N	12	12	12	12	12
TDS	Pearson Correlation	0.368	0.999**	1	0.998**	0.628*
	Sig. (2-tailed)	0.239	0.000		0.000	0.029
	N	12	12	12	12	12
Salinity	Pearson Correlation	0.371	0.999**	0.998**	1	0.661*
	Sig. (2-tailed)	0.235	0.000	0.000		0.019
	N	12	12	12	12	12
Turbidity	Pearson Correlation	0.017	0.655*	0.628*	0.661*	1
	Sig. (2-tailed)	0.958	0.021	0.029	0.019	
	N	12	12	12	12	12
**. Correlation is significant at the 0.01 level (2-tailed).						
*. Correlation is significant at the 0.05 level (2-tailed).						

In summary, the correlation analysis highlights the strong interconnectedness of EC, TDS, and salinity as indicators of dissolved substances in water, while turbidity shows a moderate association with these parameters. Meanwhile, pH appears to vary

independently, emphasizing the complexity of water chemistry and the importance of examining multiple parameters to understand water quality dynamics fully. The partial correlation analysis was conducted to assess the relationships among the water quality parameters—pH, Electrical Conductivity (EC), Total Dissolved Solids (TDS), Salinity, and Turbidity—while statistically controlling for the potential confounding effect of seasonal variation (Table 4.48). By accounting for seasonal influences, this analysis clarifies the intrinsic relationships between the variables independent of seasonal changes.

The results in Table 4.35 show that the correlations involving pH and the other parameters remain weak and statistically non-significant after controlling for seasons. This indicates that the limited relationship between pH and water quality measures is consistent regardless of seasonal variation. EC, TDS, and salinity exhibit robust positive correlations with each other, with coefficients close to perfect correlation. These relationships remain highly significant ($p < 0.001$), reinforcing that these variables are closely linked due to their mutual reflection of dissolved ionic content in the water, unaffected by seasonal changes.

Regarding turbidity, the partial correlations with EC, TDS, and salinity remain statistically significant but show a slight reduction in strength compared to the zero-order (uncontrolled) correlations. For example, the partial correlation between turbidity and EC is 0.659 ($p = 0.028$), marginally higher than the previously observed zero-order correlation of 0.655. This slight change suggests that while seasonal factors may influence some variability in turbidity and its association with other parameters, the core relationship between turbidity and dissolved solids remains robust. The partial correlation analysis confirms that seasonal variation does not substantially alter the fundamental interrelationships among EC, TDS, salinity, and turbidity. At the same time, the weak association of pH with other parameters remains unchanged. This strengthens the interpretation that EC, TDS, and salinity are intrinsically interconnected water quality indicators, and turbidity maintains a moderate but significant association with these measures regardless of seasonal fluctuations.

Table 4.35: Correlation analysis of water quality parameters by seasons

Seasonal Correlations on water quality parameters							
Control Variables			Ph	EC	TDS	Salinity	Turbidity
Seasons	pH	Correlation	1.000	0.375	0.383	0.379	0.053
		Significance (2-tailed)	0.000	0.256	0.245	0.251	0.876
		Df	0	9	9	9	9
	EC	Correlation	0.375	1.000	0.999	0.999	0.659
		Significance (2-tailed)	0.256	0.000	0.000	0.000	0.028
		Df	9	0	9	9	9
	TDS	Correlation	0.383	0.999	1.000	0.998	0.633
		Significance (2-tailed)	0.245	0.000	0.000	0.000	0.037
		Df	9	9	0	9	9
	Salinity	Correlation	0.379	0.999	0.998	1.000	0.669
		Significance (2-tailed)	0.251	0.000	0.000	0.000	0.024
		Df	9	9	9	0	9
	Turbidity	Correlation	0.053	0.659	0.633	0.669	1.000
		Significance (2-tailed)	0.876	0.028	0.037	0.024	0.000
		Df	9	9	9	9	0

When controlling for the effect of Sampling Stations, the correlation patterns among the water quality parameters remain broadly consistent with the initial zero-order correlations (Table 4.36). Specifically, the interrelationships between Electrical Conductivity (EC), Total Dissolved Solids (TDS), and Salinity continue to be extremely strong and statistically significant, with correlation coefficients close to perfect ($r^2 = 0.999$, $p < 0.001$). This reaffirms that these parameters are closely linked and likely reflect the same underlying chemical characteristics of the water regardless of sampling location.

Interestingly, after accounting for differences between sampling stations, the pH correlations show slight increases in strength. Notably, the correlation between pH and EC rises to $r^2 = 0.592$ ($p = 0.055$) and between pH and TDS to $r^2 = 0.613$ ($p = 0.045$), with the latter reaching a statistically significant level at the 0.05 threshold. These findings suggest that differences in sampling stations may partially explain some

variation in pH related to dissolved solids and that controlling for location reveals a closer relationship than initially observed.

Turbidity maintains moderately strong and significant correlations with EC, TDS, and Salinity after controlling for sampling stations. For example, the correlation between EC and Turbidity increases slightly to $r^2 = 0.682$ ($p = 0.021$), indicating a robust positive association independent of sampling location. This implies that turbidity's relationship with these water chemistry parameters is stable across different points along the stream.

Table 4.36: Correlation analysis of water quality parameters by sampling stations

Control Variables			pH	EC	TDS	Salinity	Turbidity
Sampling Station	pH	Correlation	1.000	0.592	0.613	0.594	-0.006
		Significance (2-tailed)	0.000	0.055	0.045	0.054	0.986
		Df	0	9	9	9	9
	EC	Correlation	0.592	1.000	0.999	0.999	0.682
		Significance (2-tailed)	0.055	0.000	0.000	0.000	0.021
		Df	9	0	9	9	9
	TDS	Correlation	0.613	0.999	1.000	0.998	0.657
		Significance (2-tailed)	0.045	0.000	0.000	0.000	0.028
		Df	9	9	0	9	9
	Salinity	Correlation	0.594	0.999	0.998	1.000	0.685
		Significance (2-tailed)	0.054	0.000	0.000	0.000	0.020
		Df	9	9	9	0	9
	Turbidity	Correlation	-0.006	0.682	0.657	0.685	1.000
		Significance (2-tailed)	0.986	0.021	0.028	0.020	0.000
		Df	9	9	9	9	0

Controlling for sampling stations highlights subtle but essential nuances in the relationships among water quality parameters, particularly strengthening the pH links and confirming the persistent strong associations among EC, TDS, Salinity, and Turbidity. The partial correlation analysis controlling for Sampling Levels—specifically the upstream, midstream, and downstream locations—reveals a somewhat different pattern in the relationships among the water quality parameters, as shown in Table 4.37. When accounting for the variation caused by these distinct sampling zones, the correlations between pH and the other variables shift to negative values. However, these correlations remain weak and statistically non-significant. This indicates that once the spatial differences between sampling levels are considered, pH does not have a meaningful linear association with EC, TDS, Salinity, or Turbidity.

Meanwhile, the strong interrelationships among EC, TDS, and Salinity persist but slightly reduce their strength. The correlation coefficients among these three variables decrease marginally, ranging from 0.996 to 0.997, yet remain highly significant ($p < 0.001$). This continued strong association suggests that these parameters are intrinsically linked due to their shared influence by dissolved ions in the water, regardless of sampling level.

However, Turbidity's correlations with EC, TDS, and Salinity weaken considerably after controlling for sampling levels. For example, the correlation between Turbidity and EC drops to 0.513 ($p = 0.106$), and the correlation between Turbidity and TDS falls to 0.464 ($p = 0.150$), both statistically non-significant. This notable decrease implies that the location along the stream plays a significant role in moderating the relationship between Turbidity and the other water quality variables. In other words, the effect of Turbidity on, or its association with, EC, TDS, and Salinity varies across upstream, midstream, and downstream sampling sites. Once these spatial differences are accounted for, the correlations weaken.

Table 4.37: Correlation analysis of water quality parameters by sampling levels

Correlations analysis by sampling levels							
Control Variables			pH	EC	TDS	Salinity	Turbidity
Sampling level	pH	Correlation	1.000	-0.287	-0.291	-0.274	-0.403
		Significance (2-tailed)	0.000	0.392	0.385	0.416	0.219
		Df	0	9	9	9	9
	EC	Correlation	-0.287	1.000	0.997	0.996	0.513
		Significance (2-tailed)	0.392	0.000	0.000	0.000	0.106
		Df	9	0	9	9	9
	TDS	Correlation	-0.291	0.997	1.000	0.993	0.464
		Significance (2-tailed)	0.385	0.000	0.000	0.000	0.150
		Df	9	9	0	9	9
	Salinity	Correlation	-0.274	0.996	0.993	1.000	0.527
		Significance (2-tailed)	0.416	0.000	0.000	0.000	0.096
		Df	9	9	9	0	9
	Turbidity	Correlation	-0.403	0.513	0.464	0.527	1.000
		Significance (2-tailed)	0.219	0.106	0.150	0.096	0.000
		Df	9	9	9	9	0

4.10 Summary

This chapter has exclusively discussed research findings as per the objectives. It has clearly outlined the analysed data in relation to the recommended WHO and US EPA limits of both the nutrients and elemental analysis results in drinking waters. It has also discussed the analysed water quality parameters.

CHAPTER FIVE

CONCLUSION AND RECOMMENDATIONS

5.1 Conclusion

5.1.1 Summary of the Hypotheses and Conclusions

Hypothesis 1: Farmers in Murang'a County do not have sufficient knowledge of the effects of catchment land use on wetland water quality.

This study aimed to evaluate the knowledge and attitudes of farmers in Murang'a County concerning how their land use practices impact the water quality of adjacent wetlands. Understanding the knowledge levels of local farmers is crucial, as agricultural activities are often a primary source of nutrient and sediment runoff affecting wetland ecosystems.

The survey results demonstrated a mixed range of understanding among the farmers, highlighting significant variability in their responses. The low Cronbach's alpha value of 0.197 for the knowledge-related items suggests that the questionnaire may have failed to capture a unified construct of farmer knowledge reliably, or it indicates inconsistent knowledge levels among the respondents. This low internal consistency might also reflect diverse experiences or a partial understanding of environmental impacts within the farming community.

Moreover, non-significant negative correlations between specific survey items point to misconceptions or conflicting beliefs regarding how catchment land use affects wetland ecosystems. Such misunderstandings can hinder practical conservation efforts, as farmers may underestimate the consequences of fertilizer runoff, soil erosion, or deforestation on water quality. These conflicting responses further support the hypothesis that farmers lack comprehensive knowledge about the environmental repercussions of their land management decisions.

Given these findings, Hypothesis 1 is supported: farmers in Murang'a County do not possess sufficient knowledge regarding the effects of their catchment land use on wetland water quality. This knowledge gap highlights the urgent need for targeted educational programs and extension services that focus on sustainable land use practices, environmental stewardship, and the conservation of wetland ecosystems.

The study's results confirm a significant knowledge deficit among farmers in Murang'a County about how their land use impacts wetland water quality. Addressing this deficit through community-based education, training, and participatory approaches will be critical for promoting sustainable agriculture and safeguarding wetland ecosystems for future generations.

Hypothesis 2: There are no differences in the level of nutrients, heavy metals, and pesticide inflows on small wetland ecosystems in Murang'a County.

The second hypothesis aimed to assess whether variations exist in the concentrations of key pollutants, specifically nutrients (such as nitrate and phosphate compounds), heavy metals, and pesticides (notably organophosphates like Malathion) within small wetland ecosystems across different land use types in Murang'a County. This was achieved through systematic sampling and chemical analysis of water and sediment samples from multiple wetland sites representative of diverse land uses such as agricultural, residential, and forested areas.

To statistically evaluate the differences in pollutant concentrations, the study employed Analysis of Variance (ANOVA) followed by post hoc tests to identify specific differences between land use categories. The results revealed significant variation in nutrient and heavy metal concentrations associated with land use patterns. Wetlands adjacent to intensive agricultural zones exhibited elevated nutrient levels, particularly nitrates and phosphates, attributable to the heavy application of synthetic fertilizers

Similarly, heavy metal concentrations were significantly higher in wetland points where farming activities were conducted. The presence of elevated heavy metals such as lead and cadmium in agricultural runoff may be linked to the use of inorganic fertilizers and use of organophosphate agrochemicals.

Pesticide analysis demonstrated significant spatial variability, especially concerning Malathion, a commonly used organophosphate insecticide in the region. The concentrations varied according to differing pesticide application rates and timing in the catchment, with agricultural wetlands showing the highest levels. These findings underscore how pesticide runoff directly impacts wetland water quality and highlight

the variability driven by farming practices. The statistically significant differences across nutrients, heavy metals, and pesticides corroborate the second hypothesis, affirming that land management directly influences pollutant loading and wetland health.

The study validates the second hypothesis by demonstrating that nutrient, heavy metal, and pesticide inflows vary significantly across wetlands in Murang'a County, depending on surrounding land use. This highlights the crucial role of sustainable land management and pollution control strategies to minimize adverse impacts on wetland water quality and preserve ecosystem functions. Implementing best management practices (BMPs) such as buffer strips, controlled fertilizer application, and integrated pest management could mitigate pollutant runoff and protect these vulnerable ecosystems.

Hypothesis 3: Land use systems do not affect water quality parameters (pH, EC, TDS, salinity, and turbidity).

Hypothesis H₀₃ was formulated to assess whether various land use practices influence key water quality parameters, including pH, Electrical Conductivity (EC), Total Dissolved Solids (TDS), salinity, and turbidity in wetlands of Murang'a County.

The study's design, however, emphasized analyzing pollutant inflows—specifically nutrients, heavy metals, and pesticides—and their relationship with land use. While these pollutants directly affect water quality, the research did not comprehensively or systematically measure how land use impacted the broader water quality parameters listed in the hypothesis. This represents a notable gap.

The data collected in this study did not provide direct measurements or robust statistical analysis linking land use to changes in water quality parameters. The absence of such focused data means that the third hypothesis, stating no effect of land use on water quality parameters, could neither be definitively confirmed nor rejected. This limitation highlights the need for future studies to integrate direct monitoring of these parameters across different land use types to draw more precise conclusions.

Due to the study's emphasis on pollutant inflows rather than direct water quality parameter measurements, third hypothesis remains inconclusive. This study has

insufficient evidence to support or refute the claim that land use systems do not affect pH, EC, TDS, water temperature, and turbidity. Future research should incorporate comprehensive water quality monitoring alongside land use characterization to elucidate these relationships more clearly. Such research is essential to inform integrated watershed management and conservation strategies.

5.1.2 Summary of the Research Objectives and Conclusions

Objective i: Establish the status of farmers' knowledge and attitudes about the effects of catchment land use on wetland water quality in Murang'a County, Kenya.

The study employed descriptive statistics and reliability analyses to assess the level and variability of farmers' knowledge and attitudes concerning how their land use practices affect wetland water quality. The findings revealed considerable variation in responses, indicating heterogeneous understanding among farmers in the study area.

A critical measure of internal consistency, Cronbach's alpha, yielded a low value of 0.197. This low-reliability coefficient suggests two possible interpretations:

1. **Inconsistent knowledge or understanding:** Farmers may have divergent or conflicting perceptions about the relationship between catchment land use and wetland water quality, reflecting limited or uneven awareness.
2. **Survey instrument limitations:** The survey items may not have been well-defined or sufficiently comprehensive to accurately capture the nuances of farmers' knowledge and attitudes, possibly causing confusion or varied interpretations among respondents.

Further examination of the correlation patterns among survey items supports this interpretation. While some positive correlations indicated coherence in certain aspects of farmers' responses, non-significant negative correlations point to inconsistencies or misunderstandings in how specific questions were perceived or answered.

These findings indicate that although some farmers possess basic awareness, overall knowledge about the environmental impacts of land use on wetlands remains insufficiently developed. This conclusion aligns with the tested hypothesis Ho1: that

farmers in Murang'a County do not have sufficient knowledge of the effects of catchment land use on wetland water quality.

The apparent knowledge gap underscores an urgent need for **targeted educational and extension interventions** to improve farmers' understanding of sustainable land management practices. By enhancing awareness, such programs could foster better stewardship of wetland ecosystems, ultimately contributing to improved water quality and ecosystem health in Murang'a County.

Objective ii: Examine the effects of catchment land use on nutrient, heavy metals, and pesticide inflows on Wetlands in Murang'a County, Kenya.

Analysis and Findings: The study comprehensively assessed how different catchment land use types influence the inflow of key pollutants, nutrients, heavy metals, and pesticides into wetland ecosystems. The study investigated differences across various land use categories using statistical analyses such as Analysis of Variance (ANOVA) and subsequent post hoc comparisons.

Key findings include:

- (i) **Nutrient Inflows:** Areas dominated by intensive agricultural activity exhibited significantly elevated nutrient concentrations. This is likely attributable to the extensive use of synthetic fertilizers, which contribute to nutrient runoff during precipitation events or irrigation.
- (ii) **Heavy Metals:** Similar patterns emerged for heavy metals, with higher concentrations detected in wetlands adjacent to agricultural and peri-urban land uses. This may reflect the influence of agrochemicals, soil erosion, and other anthropogenic activities within these catchments.
- (iii) **Pesticides:** Pesticide concentrations, particularly of Malathion, showed significant variability correlated with land use practices. Agricultural zones, where pesticide application is common, had higher pesticide inflows than less disturbed or natural land use areas. This variability underscores the impact of farming practices on wetland contamination.

The statistical significance of these variations, confirmed through ANOVA and post hoc analyses, supports the assertion that catchment land use is a critical determinant of pollutant inflows into wetlands. However, it is essential to note that the influence was not uniform for every parameter, suggesting complex interactions between land use type, pollutant transport mechanisms, and local environmental conditions.

The objective of examining the effects of catchment land use on pollutant inflows was successfully met. The evidence validates the hypothesis H₀₂: significant differences in the levels of nutrients, heavy metals, and pesticide inflows into wetland ecosystems depending on land use patterns.

These findings highlight the crucial role of land management and agricultural practices in shaping wetland water quality. They emphasize the need for:

- (i) Integrated catchment management strategies that minimize pollutant runoff through best management practices (BMPs), such as buffer zones, reduced chemical inputs, and erosion control.
- (ii) Targeted monitoring and regulation of pesticide and fertilizer application within sensitive catchments to protect wetland health.
- (iii) Community engagement and farmer education programs to promote sustainable land use practices that mitigate wetland contamination risks.

Objective iii: Determine changes in water quality parameters concerning wetland catchment land use practices.

While the study extensively examined nutrient, heavy metal, and pesticide inflows as indicators of catchment land use impacts on wetland ecosystems, it did not specifically measure or report detailed changes in water quality parameters such as pH, Electrical Conductivity (EC), Total Dissolved Solids (TDS), salinity, and turbidity about different land use types. These parameters are critical indicators of water quality and can reflect the influence of land use activities by capturing variations in water's chemical composition and physical characteristics. For instance:

- (i) pH provides insight into the acidity or alkalinity of wetland waters, which can be affected by agricultural runoff or industrial discharges.

- (ii) EC and TDS indicate the concentration of dissolved ions and solids, often increasing with pollutant inflows from fertilizers or soil erosion.
- (iii) Salinity reflects salt concentrations, influencing irrigation practices and land use changes.
- (iv) Turbidity measures water clarity, often increasing with sediment runoff due to land disturbance.

However, the absence of focused data or analysis on these specific parameters means that any inferred effects of land use on water quality remain indirect or speculative within the current study.

Interpretation:

- (i) The lack of direct measurement or targeted analysis of these water quality parameters limits the ability to conclusively determine their variability or changes influenced by catchment land use.
- (ii) This gap suggests that while pollutant inflows (nutrients, metals, pesticides) are affected by land use, their translation into changes in standard water quality parameters is not established in this study.
- (iii) Therefore, hypothesis H₀₃ — which posits no effects of land use systems on water quality parameters — cannot be fully confirmed or rejected.

The hypothesis H₀₃ remains inconclusive based on the available data. Although catchment land use influences pollutant inflows into small wetlands, the study lacks specific evidence to confirm or deny corresponding changes in water quality parameters such as pH, EC, TDS, salinity, and turbidity.

The general objective was to determine the impacts of catchment land use on the water quality of small wetlands in Murang'a County, Kenya.

The study effectively addressed the general objective by providing comprehensive insights into how catchment land use influences the water quality of small wetlands within Murang'a County. Specifically, the research demonstrated that different land use practices, particularly agricultural and urban activities, significantly affect the

inflows of key pollutants such as nutrients, heavy metals, and pesticides into these wetland ecosystems.

Through quantitative analyses, including ANOVA and post hoc testing, the study established that land use is a critical determinant of pollutant load variations, with intensified agricultural activities correlating with elevated nutrient and pesticide concentrations. The detection of heavy metals further underscored the anthropogenic impacts linked to land management and urban runoff within the catchment.

These findings provide an essential empirical foundation for understanding the pathways through which land use affects wetland water quality. This knowledge is critical for informing conservation policies, sustainable land management practices, and targeted interventions aimed at protecting and rehabilitating wetland ecosystems in the region.

The general objective was achieved mainly as the study successfully identified and characterized the impacts of catchment land use on water quality entering small wetlands. The results highlight the need for integrated catchment management strategies that address land use practices to safeguard wetland health and ecosystem services in Murang'a County.

5.2 Recommendations

5.2.1 Recommendations for Policy

Based on the findings of this study, the following recommendations are proposed:

(i) Collaborative Management of Wetland Water Quality

It is recommended that regular and systematic monitoring and assessment activities be conducted in collaboration with key regulatory and environmental agencies such as the National Environment Management Authority (NEMA) and the Water Resources Authority (WRA). These efforts should specifically target tracking nutrient (e.g., phosphates and nitrates) and heavy metal (e.g., cadmium, copper, lead, manganese, zinc) concentrations within the wetlands of Murang'a County.

(ii) Educational and outreach programs

Sensitisation forums should be planned and delivered to farmers, industrial stakeholders, and local communities, emphasizing the direct linkages between land use decisions and wetland water quality. Such programs could include workshops, demonstration projects, and community engagement initiatives to foster sustainable behaviors that minimize pollutant loading into wetlands. Through these integrated approaches, it is anticipated that pollutant inflows into wetlands will be significantly reduced, supporting the long-term health and ecological integrity of Murang'a County's wetland ecosystems.

(iii) Recommendations for Pesticide Management and Monitoring

In light of the analysis of pesticide concentrations within the wetland ecosystems of Murang'a County, remarkably the observed variability and potentially high concentrations of Malathion, it is imperative to implement enhanced environmental management strategies tailored to pesticide control and monitoring. A key recommendation is establishing a rigorous, systematic monitoring program targeting Malathion levels. To capture spatial variability, this should include regular and comprehensive sampling protocols conducted across multiple strategic locations within the wetlands.

5.2.2 Recommendations for Future Studies:

- (i) Future studies should incorporate systematic, spatially and temporally resolved measurements of water quality parameters across different land use types to fill this critical knowledge gap.
- (ii) Employing continuous monitoring and robust statistical analyses (e.g., multivariate approaches, time series analysis) would help elucidate the direct effects of land use on wetland water quality.
- (iii) Integrating these measurements with pollutant inflow data can provide a comprehensive understanding of how land use practices impact wetland ecosystem health.
- (iv) Development and Implementation of Pollution Control Strategies to Improve Wetland Water Quality
- (v) Further research be conducted across additional wetland ecosystems within the broader region. Such comparative studies would help determine

whether the trends observed in Murang'a County are consistent across other catchments with varying land use patterns and environmental conditions. Moreover, future studies should consider increasing both the sample size and the frequency of monitoring events.

(vi) Recommendations for Further Research on Thiamethoxam Distribution:

The significant difference observed in Thiamethoxam concentrations between sediment and water samples highlights the need for a deeper investigation into this pesticide's environmental behavior and distribution patterns within wetland ecosystems. To address this, it is recommended that future studies undertake detailed research to understand the mechanisms driving Thiamethoxam's varying presence in different environmental compartments.

(vii) Further research should be carried out to ascertain the existence of Lead (Pb) in Murang'a wetland waters and investigate other potential sources apart from the associated agricultural sources.

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APPENDICES

Appendix 1: Postgraduate Approval for Research



MURANG'A UNIVERSITY OF TECHNOLOGY

Office of the Director, Postgraduate Studies

E-mail: hps@mut.ac.ke

Website: www.mut.ac.ke

P.O. Box 75-10200

Murang'a, Kenya

Ref: MUT/ARP/PGS/19/2020 /VOL.1

Date: 7th November, 2023

Mr. Kipkemoi Evans (AS501/5726/2021)
School of Pure, Applied and Health Sciences,
Murang'a University of Technology,
P.O. Box 75-10,200
Murang'a.

Dear Mr. Kipkemoi,

RE: APPROVAL OF RESEARCH PROPOSAL AND SUPERVISORS.

I am pleased to inform you that the Directorate of Postgraduate Studies on 13th October, 2023 considered and approved your Doctoral research proposal entitled "*Impacts of Catchment Land Use on Water Quality of Small Wetlands in Murang'a County, Kenya*" and appointed the following as supervisors:

1. Dr. Warren A. Andayi - Murang'a University of Technology
2. Prof. Eric Chomba Njagi, PhD - Chuka University
3. Prof. Benson M. Mwangi, PhD - Murang'a University of Technology

You may now proceed with your data collection subject to obtaining research permit from NACOSTI, if required. You should also begin consulting your supervisors and submit through them quarterly progress reports to the Director Postgraduate Studies through your CoD and School Dean. Progress Reports forms can be accessed in the Student Portal.

It is the policy and regulations of the University that you observe deadlines. The guidelines on Postgraduate supervision can be accessed in the post graduate Handbook.

Your responsibilities as a student will include, among others;

- i. Maintain regular consultation with your supervisor(s), at least once a month
- ii. Submit quarterly reports on time, through your supervisors, CoD, Dean and to the Director of Postgraduate Studies;
- iii. Ensure quality work all through;



MUT IS ISO 9001: 2015 & ISO/IEC 27001:2013 CERTIFIED



- iv. Present your research findings at 2-3 seminars/conferences prior to thesis examination.
- v. Publish two articles from your research findings in a refereed journal prior to thesis examination.

For any further clarification please contact the undersigned.

Yours Sincerely,



Prof. Geoffrey Muchiri, PhD

Director, Postgraduate Studies

GM/PIM



Copy to:

1. Registrar – Academics and Students Affairs
2. Dean, School of Pure, Applied and Health Sciences

Appendix 2: Research Permit (NACOSTI)

 <p>REPUBLIC OF KENYA</p> <p>Ref No: 441382</p>	 <p>NATIONAL COMMISSION FOR SCIENCE, TECHNOLOGY & INNOVATION</p> <p>Date of Issue: 09/June/2025</p>
<p>RESEARCH LICENSE</p>	
	
<p>This is to Certify that Mr. EVANS KIPKEMOI of Murang'a University of Technology, has been licensed to conduct research as per the provision of the Science, Technology and Innovation Act, 2013 (Rev.2014) in Muranga on the topic: IMPACTS OF CATCHMENT LAND USE ON WATER QUALITY STATUS OF SMALL WETLANDS IN MURANG'A COUNTY, KENYA for the period ending : 09/June/2026.</p>	
<p>License No: NACOSTI/P/25/4174888</p>	
<p>441382</p>	
<p>Applicant Identification Number</p>	
<p>Deputy Director</p>	
<p>NATIONAL COMMISSION FOR SCIENCE, TECHNOLOGY & INNOVATION</p>	
<p>Verification QR Code</p>	
	
<p>NOTE: This is a computer generated License. To verify the authenticity of this document, Scan the QR Code using QR scanner application.</p>	
<p>See overleaf for conditions</p>	

Appendix 3: Questionnaire on Knowledge and Attitude

Part one: Knowledge

Directions: Please indicate your agreement or disagreement with these statements regarding wetlands near you. Place an "X" mark in the box to signify your answer.

1: I can identify wetlands in my neighborhood

☐ **Yes** ☐ **No**

2: Wetlands are important to me directly

☐ **Yes** ☐ **No** ☐ **I do not know**

3: Wetlands are important to all of us

☐ **Yes** ☐ **No**

4: Has the size of the wetland around decreased?

☐ **Yes** ☐ **No** ☐ **I do not know**

5: Wetlands drain into the River Mathioya

☐ **Yes** ☐ **No** ☐ **I do not know**

6: Cutting trees have negatively impacted on wetlands

☐ **Yes** ☐ **No** ☐ **I do not know**

7: The amount of agro-chemicals used can be harmful to wetlands

☐ **Yes** ☐ **No** ☐ **I do not know**

8. The type of agrochemicals used can be harmful to wetlands

☐ **Yes** ☐ **No** ☐ **I do not know**

9. The method of applying agrochemicals can be harmful to wetlands

☐ **Yes** ☐ **No** ☐ **I do not know**

10. Farmers are aware of safe agricultural practices

☐ **Yes** ☐ **No** ☐ **I do not know**

Part Two: Attitude

11. Farmers hardly care about the existence of wetlands

☐ **Yes** ☐ **No**

12. Farmers hardly care about the type of agrochemicals they use in their farms

Yes ☐ ☐ **No**

13. Farmers are hardly concerned about the conservation of wetlands

Yes ☐ **No**

14. Farmers are hardly concerned about the safety of methods they use in applying agrochemicals in the farms

☐ **Yes** ☐ **No**

15. Farmers hardly find value in attending conservation awareness forums

☐ **Yes** ☐ **No**

16. Farmers hardly apply conservation methods they are aware of

☐ **Yes** ☐ **No**

17. Farmers hardly find the need to observe safety regulations when applying chemicals to the farm

☐ **Yes** ☐ **No**

18. Farmers hardly care about the effects of agrochemicals they use on the farm on wetlands

☐ **Yes** ☐ **No**

19. Farmers are more concerned with the yields they get from the farm than the safety of wetlands

☐ **Yes** ☐ **No**

20. Pollution of wetlands is not a concern for farmers

☐ **Yes** ☐ **No**

Appendix 4: Pesticide Inflows Chromatograms



<Sample Information>

Sample Name: Malathion

Sample ID: 20 PPM (std)

Data Filename: 0.008 ppm.lcd

Method File name: malathion.lcm

Batch Filename: AA 2024 GROUP.lcb

Vial #: 1-2 Sample Type: Standard

Injection Volume: 4 uL

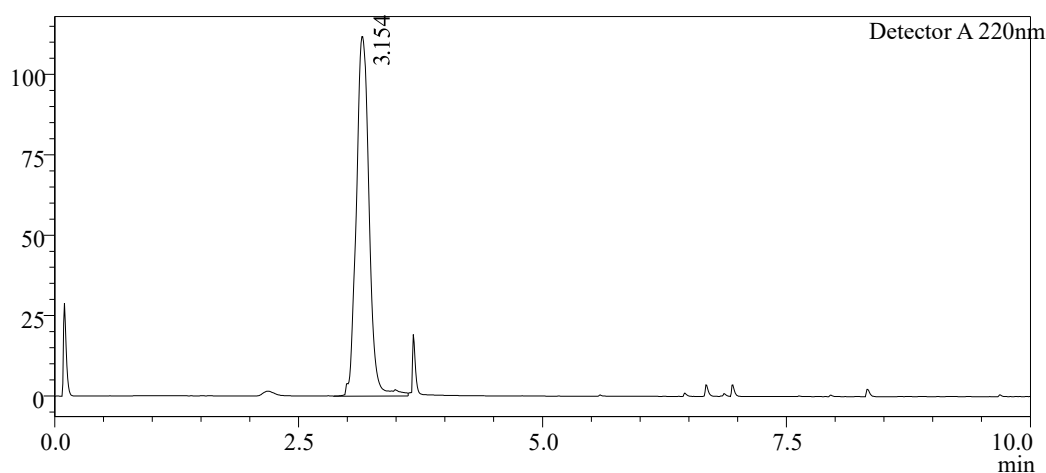
Level: 1

Date Acquired : 11/20/2024 9:34:11 PM Acquired by :

System Administrator Date Processed : 11/20/2024 11:00:32 PM

Processed by : System Administrator

<Chromatogram> mV



<Peak Table>

Detector A 220nm

Peak#	Ret. Time	Area	Height	Conc.	Unit	Mark	Name
1	3.154	1011436	111779	0.008	mg/L		malathion
Total		1011436	111779				

<Sample Information>

Sample Name: Thiamethoxam

Sample ID: 0.01016 PPM (std)

Data Filename: 0.01016 PPM_5.lcd

Method File name: Thiamethoxam.lcm

Batch Filename: malathion.lcb

Vial #: 1-2 Sample Type: Standard

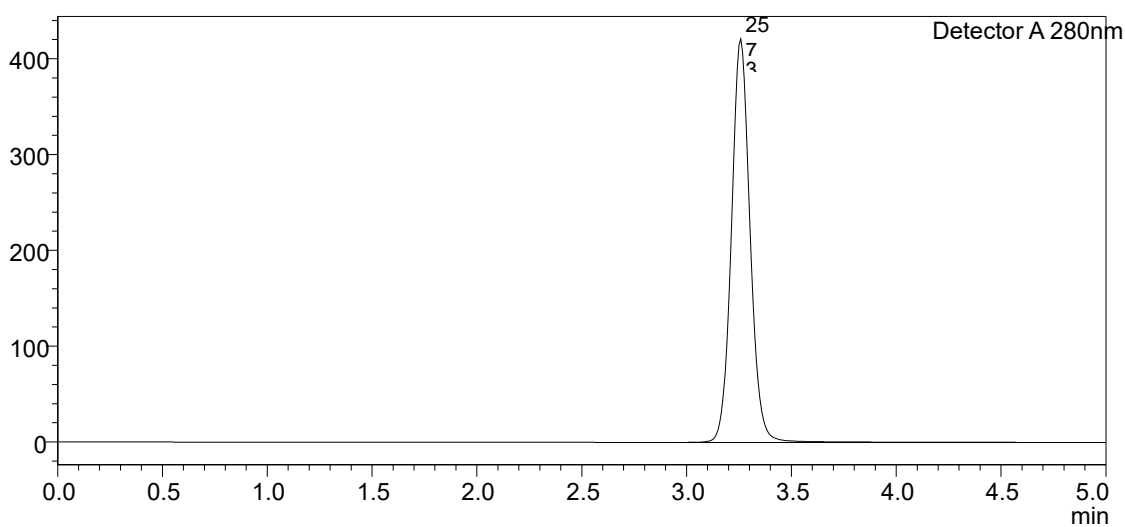
Injection Volume: 4 uL Level: 1

Date Acquired : 11/21/2024 8:37:42 PM Acquired by : System Administrator

Date Processed : 11/21/2024 8:52:16 PM Processed by : System

Administrator

<Chromatogram> mV



<Peak Table>

Detector A 280nm

Peak#	Ret. Time	Area	Height	Conc.	Unit	Mark	Name
1	3.257	2622224	421076	0.010	mg/L	S	Thiamethoxam
Total		2622224	421076				

<Sample Information>

Sample Name : Diazinon

Sample ID: 0.01036 PPM (std)

Data Filename: 0.01036 PPM_2.lcd

Method File name: Diazinon.lcm

Batch Filename: malathion.lcb

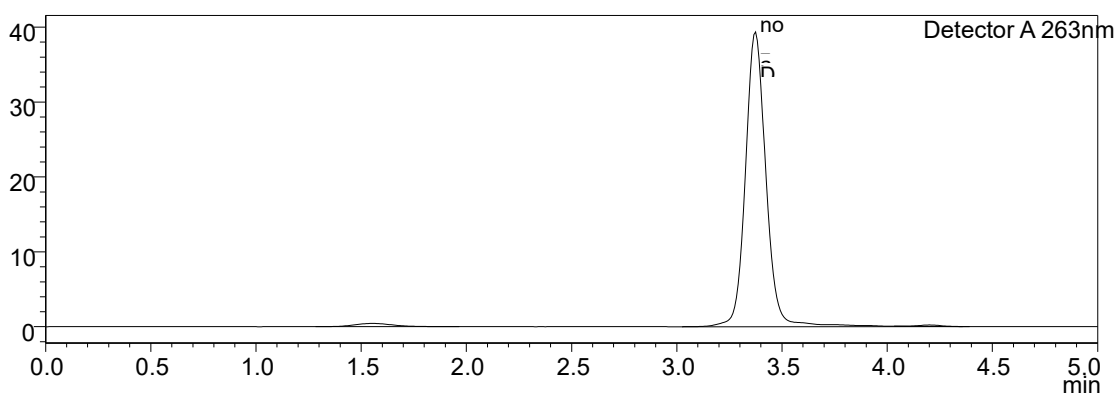
Vial #: 1-2 Sample Type: Standard

Injection Volume: 4 uL Level: 1

Date Acquired : 11/23/2024 6:50:07 PM Acquired by : System Administrator

Date Processed : 11/23/2024 7:02:06 PM Processed by : System Administrator

<Chromatogram> mV



<Peak Table>

Detector A 263nm

Peak#	Ret. Time	Area	Height	Conc.	Unit	Mark	Name
1	1.554	5542	435	0.000			
2	3.373	268247	39356	0.010	mg/L	S	Diazinon
3	4.198	1329	185	0.000		T	
Total		275118	39976				

<Sample Information>

Sample Name: Malathion

Sample ID: 1 b

Data Filename: 1 b.lcd

Method Filename: malathion.lcm

Batch Filename: AA 2024 GROUP.lcb

Vial #: 1-3 Sample Type: Unknown

Injection Volume: 20 uL

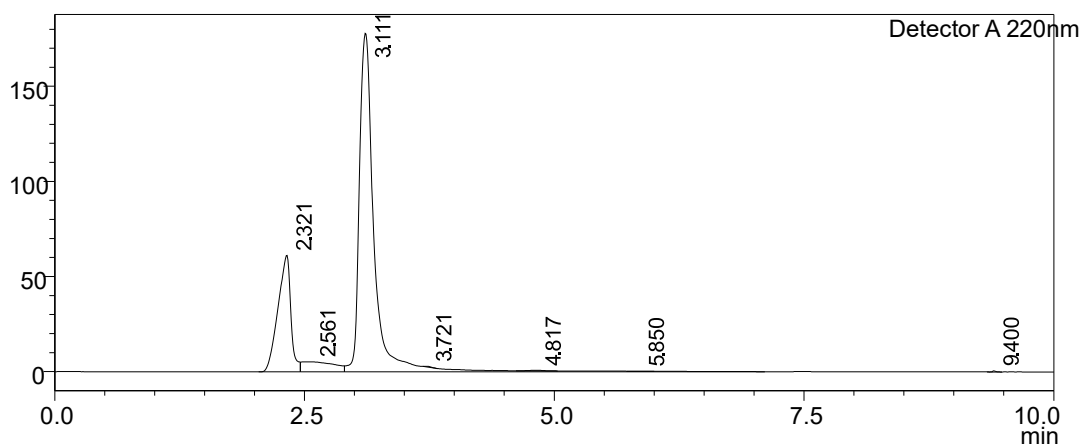
Level: 1

Date Acquired : 11/20/2024 10:00:58 PM Acquired by :

System Administrator Date Processed : 11/20/2024 10:10:59 PM

Processed by : System Administrator

<Chromatogram> mV



<Peak Table>

Detector A 220nm

Peak#	Ret. Time	Area	Height	Conc.	Unit	Mark	Name
1	2.321	545685	61251	0.000			
2	2.561	119854	5253	0.000		V	
3	3.111	1841402	177939	0.014	mg/L	SV	malathion
4	3.721	1640	353	0.000		T	
5	4.817	3154	348	0.000		T	
6	5.850	1063	76	0.000		TV	
7	9.400	1233	601	0.000		V	
Total		2514031	245821				

<Sample Information>

Sample Name : Diazinon

Sample ID: 1b Wet

Data Filename: 1 b_1.lcd

Method Filename: Diazinon.lcm

Batch Filename: malathion.lcb

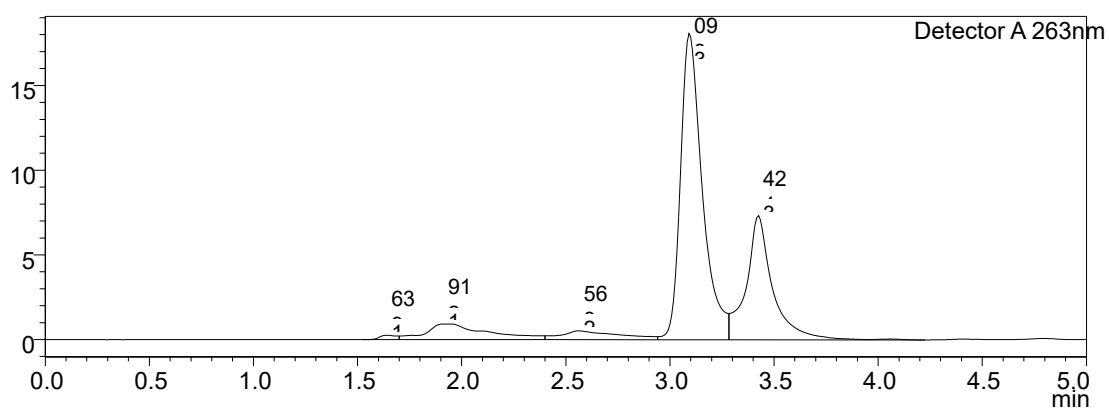
Vial #: 1-3 Sample Type: Unknown

Injection Volume: 20 uL Level: 1

Date Acquired : 11/23/2024 7:04:01 PM Acquired by : System Administrator

Date Processed : 11/23/2024 7:09:02 PM Processed by : System Administrator

<Chromatogram> mV



<Peak Table>

Detector A 263nm

Peak#	Ret. Time	Area	Height	Conc.	Unit	Mark	Name
1	1.639	1471	257	0.000			
2	1.912	19195	922	0.000		V	
3	2.563	10267	523	0.000		V	
4	3.093	132846	18080	0.000		V	
5	3.424	63999	7338	0.001	mg/L	SV	Diazinon
Total		227777	27120				

<Sample Information>

Sample Name: Thiamethoxam

Sample ID: 8s wet

Data Filename: 2s.lcd

Method Filename: Thiamethoxam.lcm

Batch Filename: malathion.lcb

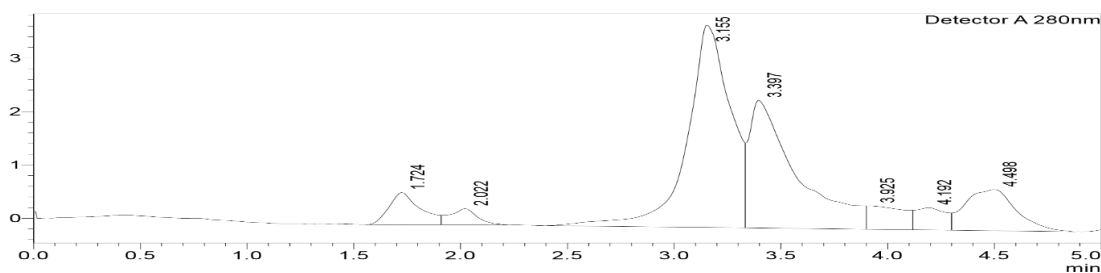
Vial #: 1-16 Sample Type: Unknown

Injection Volume: 20 uL Level: 1

Date Acquired : 11/21/2024 9:54:48 PM Acquired by : System Administrator

Date Processed : 11/21/2024 9:59:48 PM Processed by : System Administrator

<Chromatogram> mV



<Peak Table>

Detector A 280nm

Peak#	Ret. Time	Area	Height	Conc.	Unit	Mark	Name
1	1.724	6175	604	0.000			
2	2.022	2679	304	0.000		V	
3	3.155	53890	3787	-0.002	mg/L		Thiamethoxam
4	3.397	36778	2389	0.000		V	
5	3.925	5212	438	0.000		V	
6	4.192	4110	419	0.000		V	
7	4.498	12698	773	0.000		V	
Total		121542	8714				

<Sample Information>

Sample Name: Thiamethoxam

Sample ID: 8b Dry

Data Filename: 8b.lcd

Method Filename: Thiamethoxam.lcm

Batch Filename: malathion.lcb

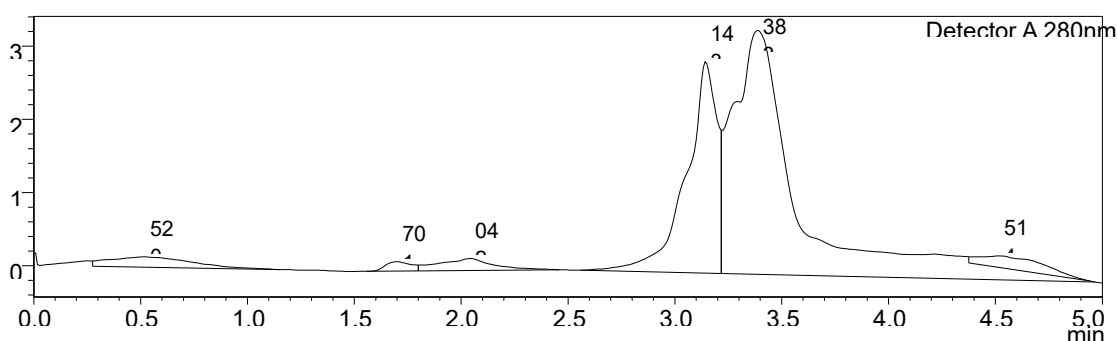
Vial #: 1-10 Sample Type: Unknown

Injection Volume: 20 uL Level: 1

Date Acquired : 11/21/2024 9:21:42 PM Acquired by : System Administrator

Date Processed : 11/21/2024 9:26:42 PM Processed by : System Administrator

<Chromatogram> mV



<Peak Table>

Detector A 280nm

Peak#	Ret. Time	Area	Height	Conc.	Unit	Mark	Name
1	0.521	3760	141	0.000		V	
2	1.701	1115	129	0.000			
3	2.043	2881	164	0.000		V	
4	3.144	29239	2888	-0.002	mg/L		Thiamethoxam
5	3.388	70672	3330	0.000		SV	
6	4.518	3565	158	0.000		TV	
Total		111232	6809				

<Sample Information>

Sample Name : Carbendazim

Sample ID: 9s wet

Data Filename: 3s.lcd

Method Filename : carbendazim.lcm

Batch Filename: malathion.lcb

Vial #: 1-17 Sample Type: Unknown

Injection Volume: 20 uL Level: 1

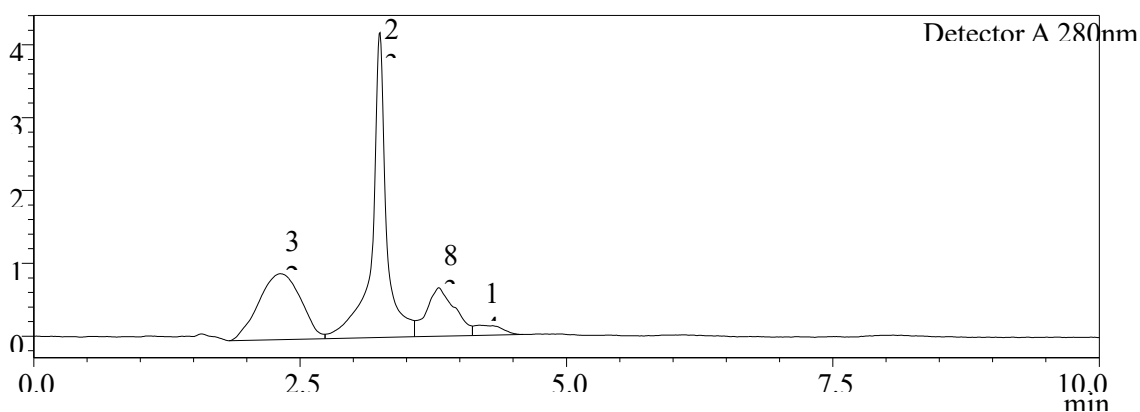
Date Acquired : 11/24/2024 9:08:21 PM

Acquired by : System Administrator

Date Processed : 11/24/2024 9:18:22 PM

Processed by : System Administrator

<Chromatogram> mV



<Peak Table>

Detector A 280nm

Peak#	Ret. Time	Area	Height	Conc.	Unit	Mark	Name
1	2.311	24799	905	0.000			
2	3.246	38971	4186	0.000		V	
3	3.801	12441	669	0.000		V	
4	4.189	2446	139	0.000		V	
Total		78658	5899				

<Sample Information>

Sample Name : Carbendazim

Sample ID: 7 b Dry

Data Filename: 7 b.lcd

Method Filename : carbendazim.lcm

Batch Filename: malathion.lcb

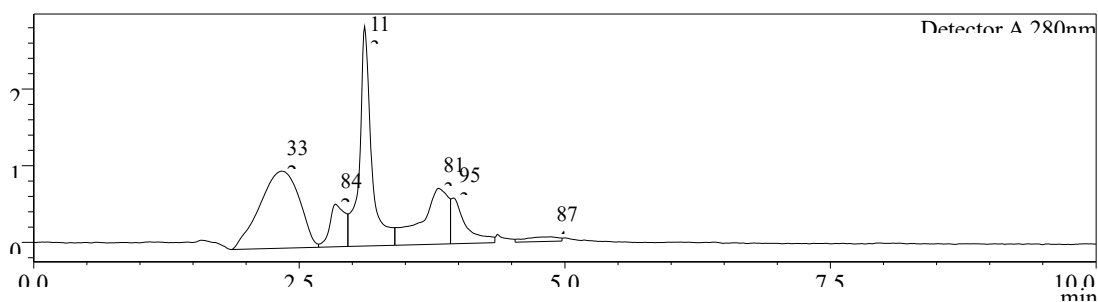
Vial #: 1-9 Sample Type: Unknown

Injection Volume: 20 uL Level: 1

Date Acquired : 11/24/2024 7:44:16 PM Acquired by : System Administrator

Date Processed : 11/24/2024 7:54:17 PM Processed by : System Administrator

<Chromatogram> mV



<Peak Table>

Detector A 280nm

Peak#	Ret. Time	Area	Height	Conc.	Unit	Mark	Name
1	2.334	26022	1005	0.000			
2	2.841	5651	557	0.000		V	
3	3.116	22619	2864	0.000		V	
4	3.812	13163	727	0.000		V	
5	3.958	5725	596	0.000		V	
6	4.873	1309	59	0.000		V	
Total		74488	5808				

Appendix 5: Evidence of Data Collection and Lab Analysis



(a) Elemental Analysis using MP-AES Instrument



(b) Packed water/sediment samples ready for lab analysis

Appendix 6: Publications

This thesis has produced the following publications:

Kipkemoi, E., Andayi, W. A., Mwangi, B. M., Njagi, E. C., & Kariuki, M. W. (2025).

Effects of Catchment Land Use on Nutrients and Heavy Metals Inflows into Maragua and Mathioya Wetlands in Murang'a County, Kenya. *International Journal of Research and Innovation in Applied Science*, 10(4), 494-506.

Kipkemoi, E., Andrew A. Andayi., B.M. Mwangi., Eric C. Njagi., & Kariuki, M. W.

(2025). Effects of Catchment Land Use on Water Quality in Maragua and Mathioya Riverine Wetlands, Murang'a County, Kenya. *International Journal of Research and Innovation in Applied Science*, 10(5), 1363-1375.