DYNAMICS IN TROPHIC STATUS, WATER AND NUTRIENT MASS BALANCE AND COMMUNITIES OF LAKE KANYABOLI, KENYA

BY

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DECLARATION

Declaration by the Candidate

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DEDICATION

To my mom (Martha Nyajoloti Muyola Kondowe), late dad (Harison Muyola Kondowe), siblings (Maggie, Gruno, Barrettie, Ellah and Siyani), late brother Titus Chimutu Muyola Kondowe and all relatives that supported me during my academic journey. Thank you, mom, and siblings, for your support, love and endless prayers. Late dad and brother Titus may your souls continue resting in peace. Unfortunately, you never lived to celebrate and enjoy the fruits of your hard work.

ABSTRACT

Fishing pressure and nutrient enrichment from the catchment area threaten Lake Kanyaboli's water quality and communities. Understanding these impacts and their effect on the lake's water quality and communities is crucial for lake management and conservation. This study assessed the spatio-temporal variations in the physical, chemical, and biological variables, nutrient budget, plankton and fish communities of Lake Kanyaboli in western Kenya from February 2020 to February 2021. Primary and secondary data on water quality variables, including Secchi depth (SD), electrical conductivity (EC), temperature, pH, dissolved oxygen (DO) concentration, nitrates (NO_3^-), nitrite (NO_2^-), ammonium (NH_4^+), soluble reactive phosphorus (SRP), total phosphorus (TP) and total nitrogen (TN), and chlorophyll-*a* (Chl-*a*) were collected. Plankton and fish community data were also collected. Water quality variables and plankton communities were compared using two-way ANOVA, PCA, Pearson correlation, and descriptive statistics. The lake’s trophic status was calculated using SD, TP and Chl-*a* Carlson trophic status index (TSI) equations. Furthermore, PERMANOVA, Two-way nested ANOSIM, species diversity indices, CCA, NMDS, Kruskal Wallis test and SIMPER were also used to compare plankton and fish community data. Mass balance models were used to calculate water, nitrogen, and phosphorus mass balances. Two-way ANOVA on water quality variables showed no site × season interactions, while only Chl-*a* showed spatial variability. Significant seasonal differences were observed in SD, DO and Chl-*a*. Based on the Chl-*a* and SD TSI values, Lake Kanyaboli is eutrophic, whereas the TP value indicates hypereutrophic conditions. The nutrient mass balance showed that the lake receives more nutrient loads of N (90.78 t N $year^{-1}$) and P (24.06 t N $year^{-1}$) than it loses for both N (73.42 t N $year^{-1}$) and P (15.76 t P $year^{-1}$). Phytoplankton consisted of 30 genera dominated by Chlorophyceae (42.30%), with Bacillariophyceae, Cyanophyceae, Cryptophyceae, Euglenoidae, Trebouxiophyceae, and Zygnematophyceae also present. Fifteen zooplankton genera were also identified, including Cladocera (16.73%), Rotifera (27.86%) and Copepoda (55.41%). Although plankton abundance differed significantly between seasons, Two-way ANOVA showed no significant site × season interaction. The 2020 fish catch survey showed that Tilapia (50%), *Clarias* sp. (23%), *Protopterus aethiopicus* (20%), and haplochromines (7%) were the main fish species in the catch. But Kruskal Wallis showed no monthly variations in fish catches. The inter-annual fish catch and catch per unit effort (2003 to 2020) showed a decreasing trend. Despite infrequent historical data, physical, chemical, and biological variables captured anthropogenic effects over time, and the lake was eutrophic due to nutrient load. The phytoplankton community also showed that the lake was in a non-equilibrium state due to short water residence time and the uptake of nutrients by macrophytes bordering the lake. The major management issues identified are nutrient loading, fishing effort, and lake connectivity with the Yala River. Managing the catchment area holistically and monitoring the lake's water quality and fishing effort will help sustain goods and services, and livelihoods of Lake Kanyaboli riparian communities.

TABLE OF CONTENTS

DECLARATION ii

DEDICATION iii

ABSTRACT...iv

TABLE OF CONTENTS v

LIST OF TABLES...xii

LIST OF FIGURES xv

LIST OF ABBREVIATIONS, ACRONYMS, AND SYMBOLS xviii

ACKNOWLEDGEMENTS xx

CHAPTER ONE 1

INTRODUCTION 1

1.1. Background 1

1.2. Statement of the problem 7

1.3. Justification of the study 12

1.4. Study objectives 14

  1.4.1. General objective 14

  1.4.2. Specific objectives 15

1.5. Study hypotheses..15

CHAPTER TWO 16

LITERATURE REVIEW 16

2.1. Limnological characteristics of lakes 16

2.1.1. Dissolved oxygen 17

2.1.2. Water transparency 19

2.1.3. Water temperature 19

2.1.4. Potential of hydrogen (pH) 21

2.1.5. Electrical conductivity 22

2.16. Chlorophyll-*a* 23

2.1.7. Nutrients 24

2.1.8. Trophic status in lakes 26

2.2. Nutrient and water mass balance in lakes 28

2.3. Phytoplankton and zooplankton communities in lakes 32

2.4. Fish and fisheries in lakes: composition, abundance, catch and effort 38

2.5. Human impact on physico-chemical variables and ecology of lakes 47

CHAPTER THREE 51

MATERIAL AND METHODS 51

3.1. Description of the study area 51

3.2. Data sources 56

3.3. Sampling sites 57

3.4. Sample collection and analysis 58

3.4.1. Objective 1: Historical changes in the limnological characteristics and trophic status of Lake Kanyaboli from 1981 to 2020 58

3.4.2. Objective 2: Factors that influence nutrient (nitrogen and phosphorus) dynamics in Lake Kanyaboli using a mass balance model 60

3.4.2.1. Sample collection 60

3.4.3. Objective 3: Spatio-temporal dynamics in environmental conditions and how they drive plankton communities in Lake Kanyaboli 64

3.4.3.1. Sample collection 64

3.4.4. Objective 4: Temporal changes in fish species composition, catch and fishing effort in Lake Kanyaboli 65

3.4.4.1. Sample collection 65

3.4. Data treatment and analysis 68

3.5.1. Objective 1: Historical changes in the limnological characteristics and trophic status of Lake Kanyaboli from 1981 to 2020 68

3.5.1.1. Trophic status 68

3.5.2. Objective 2: Factors that influence nutrient (nitrogen and phosphorus) dynamics in Lake Kanyaboli using a mass balance model 70

3.5.2.1. Nutrient load estimate...73

3.5.3. Objective 3: Spatio-temporal dynamics in environmental conditions and how they drive plankton (zooplankton and phytoplankton) communities in Lake Kanyaboli 74

3.5.3.1. Analysis of plankton species diversity 75

3.5.4. Objective 4: Temporal changes in fish species composition, catch and fishing effort in Lake Kanyaboli 75

CHAPTER FOUR...79

RESULTS 79

4.1. Objective 1: Historical changes in the limnological characteristics and trophic status of Lake Kanyaboli from 1981 to 2020 79

4.1.1. Temporal changes in the water quality of Lake Kanyaboli 79

4.1.2. Trophic status of Lake Kanyaboli..81

4.1.3. Water quality changes in Lake Kanyaboli 83

4.2. Objective 2: Factors that influence nutrient (nitrogen and phosphorus) dynamics in Lake Kanyaboli using a mass balance model 89

4.2.1. Water balance 89

4.2.2. Nutrient balance 93

4.3. Objective 3: Spatio-temporal dynamics in environmental conditions and how they drive plankton communities in Lake Kanyaboli 99

4.3.1. Phytoplankton and zooplankton community composition in Lake Kanyaboli 99

4.3.2. Influence of physico-chemical variables on plankton communities 111

4.4. Objective 4: Temporal changes in fish species composition, catch and fishing effort in Lake Kanyaboli 113

4.4.1. Characteristics of Lake Kanyaboli fishery 113

4.4.2. Historical trends in fish catches, effort and composition of Lake Kanyaboli 114

4.4.3. Temporal variations in fish composition and catches in Lake Kanyaboli from February 2020 to February 2021 118

4.4.4. Catch per unit effort in Lake Kanyaboli from February 2020 to February 2021 125

CHAPTER FIVE 127

DISCUSSION 127

5.1. Historical changes in the limnological characteristics and trophic status of Lake Kanyaboli 127

5.1.1. Trophic status 132

5.2. Water and nutrient mass balance in Lake Kanyaboli 133

5.3. Plankton communities in Lake Kanyaboli 138

5.3.1. Dynamics in phytoplankton communities 139

5.3.2. Dynamics in zooplankton communities 142

5.3.3. Seasonal variation in phytoplankton and zooplankton 144

5.4. Fish and fisheries of Lake Kanyaboli 145

5.4.1. Historical trends in fish composition, catches and effort 146

5.4.2. Fish species diversity, abundance catch and catch per unit effort changes between February 2020 and February 2021 149

5.4.3. Management implications of the fish composition and production study 151

CHAPTER SIX 155

CONCLUSIONS AND RECOMMENDATIONS 155

6. Conclusions 155

6.1.1. Historical changes in the Limnological characteristics of Lake Kanyaboli .155

6.1.2. Nutrient mass balance in Lake Kanyaboli 156

6.1.3. Phytoplankton and zooplankton communities in Lake Kanyaboli 156

6.1.4. Fishes and fisheries in Lake Kanyaboli 157

6.2. Recommendations 158

REFERENCES..160

APPENDICES 219

Appendix I: Nutrient concentration calculation equations..219

Appendix II: Levene homogeneity test for 12 selected variables measured in Lake Kanyaboli, Kenya, during February 2020 to February 2021 sampling period 222

Appendix III: Two-way ANOVA results for 12 variables measured in Lake Kanyaboli, Kenya, during February 2020 to February 2021 sampling period 223

Appendix IV: Similarity report 225

LIST OF TABLES

Table 1: General features of Lake Kanyaboli, Kanya......54

Table 2: Criteria for classification of freshwater lake trophic status......70

Table 3: Engine capacity of non-motorized fishing vessels by length class for small-scale vessels......77

Table 4: Summary of water quality in Lake Kanyaboli between 1981 and 2020 as captured in different studies......80

Table 5: Trophic status index of Lake Kanyaboli between 1981 and 2020 based on Secchi depth, total phosphorus, and chlorophyll-*a* concentrations.......82

Table 6: Mean values (±StdDev) among sampling stations in Lake Kanyaboli, Kenya, from February 2020 to February 2021. Means with different letters across the row are significantly different at $\alpha = 0.05$ as determined by multiple comparisons (one-way ANOVA followed by post hoc Tukey)......87

Table 7: Pearson correlation coefficients within various water quality variables in Lake Kanyaboli from February 2020 to February 2021......88

Table 8: Estimated hydrological parameters of Lake Kanyaboli based on the present study from February 2020 to February 2021.......91

Table 9: Estimated water budgets for Lake Kanyaboli based on the present study from February 2020 to February 2021.......92

Table 10: Seasonal variations in Lake Kanyaboli nutrient loads from the atmosphere, inflow and outlet between February 2020 and February 2021..95

Table 11: Sources of nutrient loads and nutrient losses in Lake Kanyaboli between February 2020 and February 2021. 96

Table 12: Seasonal variations in Lake Kanyaboli nutrient loads from the atmosphere, inlet canal and outlet between February 2020 and February 2021. (a): inlet canal and (b): outlet. 97

Table 13: Temporal changes in diversity indices and taxa for phytoplankton and zooplankton communities in Lake Kanyaboli from February 2020 to February 2021 100

Table 14: Mean relative abundance of phytoplankton taxa recorded in Lake Kanyaboli from February 2020 to February 2021 106

Table 15: Mean relative abundance of zooplankton taxa recorded in Lake Kanyaboli from February 2020 – February 2021...108

Table 16: Zooplankton ranked abundance SIMPER contributors to % dissimilarity in phytoplankton composition between the wet and dry seasons. 110

Table 17: Summary of characteristics of the fishing effort of Lake Kanyaboli between 2003 and 202...114

Table 18: Annual effective fishing effort, catch and catch per unit effort between 2003 and 2020 in Lake Kanyaboli...115

Table 19: Comparison of fish species between 1981 and 2020 in Lake Kanyaboli 117

Table 20: Temporal changes in diversity indices and the number of fish taxa in Lake Kanyaboli from February 2020 to February 2021. 119

Table 21: Species-wise monthly fish abundance caught in Lake Kanyaboli from February 2020 to February 2021 121

Table 22: Relative abundance (percentage, %) contribution of each fish species to the total annual abundance of all fishes caught in Lake Kanyaboli from February 2020 to February 2021. 122

Table 23: Percentage contribution of each fish species to the total annual catch from Lake Kanyaboli between February 2020 and February 2021 124

LIST OF FIGURES

Figure 1: Chronology of events in Lake Kanyaboli and Yala Wetland from 1954 to 2017. 11

Figure 2: Map of the Yala Wetland with Lakes Kanyaboli, Namboyo, and Sare (source: Abila *et al*., 2008). 52

Figure 3: Map showing six sampling sites in Lake Kanyaboli, inlet and outlet from February 2020 to February 2021 58

Figure 4: Conceptual diagram showing the water and nutrient sources and losses in Lake Kanyaboli 62

Figure 5: Water transparency in Lake Kanyaboli water transparency over time 83

Figure 6: Temporal variation in physical, chemical, and biological variables in Lake Kanyaboli from February 2020 to February 2021. SD: Secchi depth; Temp.: Temperature; EC: Electrical conductivity; DO: Dissolved oxygen; TP: Total phosphorus; TN: Total nitrogen, Chl-*a*: Chlorophyll-*a*, SRP: Soluble reactive phosphorus; NH_4^+: Ammonium; NO_3^-: Nitrate; NO_2^-: Nitrite 84

Figure 7: PCA comparison of physico-chemical variables in Lake Kanyaboli based on: (a) site (b) season from February 2020 to February 2021. KAD: Kadenge, MID: Open water 85

Figure 8: Monthly discharge rates into and out of Lake Kanyaboli from February 2020 to February 2021. 90

Figure 9: Monthly total nitrogen and phosphorus loads into and out of Lake Kanyaboli between February 2020 and February 2021. (a) = total nitrogen and (b) = total phosphorus. 94

Figure 10: Schematic diagram showing the nutrient sources and losses in Lake Kanyaboli from February 2020 to February 2021. 98

Figure 11: Comparison of mean plankton abundance at different sampling stations in Lake Kanyaboli from February 2020 to February 2021 (a: phytoplankton; b: zooplankton) 102

Figure 12: Temporal variations of phytoplankton and zooplankton abundance in Lake Kanyaboli from February 2020 to February 2021 (a: monthly variations of phytoplankton; b: monthly variations of zooplankton). 104

Figure 13: NMDS (stress value = 0.1) comparison of phytoplankton and zooplankton structure in Lake Kanyaboli from February 2020 to February 2021. (a) phytoplankton site comparisons, (b) phytoplankton seasonal comparisons, (c) zooplankton site comparisons (d) zooplankton seasonal comparisons. KAD: Kadenge, MID: Open water 109

Figure 14: Canonical correspondence analysis (CCA) between plankton and physico-chemical variable among sites in Lake Kanyaboli from February 2020 to February 2021. (a-b) phytoplankton site comparisons, (c-d) zooplankton site comparisons. KAD: Kadenge, MID: Open water 112

Figure 15: Monthly fish catch (mt) in Lake Kanyaboli from February 2020 to February 2021. 125

Figure 16: Monthly effective fishing effort and catch per unit effort in Lake Kanyaboli from February 2020 to February 2021. CPUE = catch per unit effort. 126

LIST OF ABBREVIATIONS, ACRONYMS, AND SYMBOLS

ANOSIM:	Two-way nested analysis of similarities
ANOVA:	Analysis of variance
APHA:	American Public Health Association
COTRA:	Collaborative Training in Fisheries and Aquaculture in Eastern, Central, and
CTSI:	Carlson Trophic State Index
DO:	Dissolved oxygen
FAO:	Food and Agriculture Organisation of the United Nations
KMFRI:	Kenya Marine and Fisheries Research Institute
KWS:	Kenya Wildlife Service
LVBA:	Lake Victoria Basin Authority
NH_4^+:	Ammonium
NMDS:	Non-metric multidimensional scaling
NO_2^-:	Nitrite
NO_3^-:	Nitrate
PCA:	Principal component analysis
SD:	Secchi Disk
SIMPER:	Similarity percentages analysis
	Southern Africa
SRP:	Soluble reactive phosphorus
TL:	Trophic Level
TN:	Total nitrogen

TP:	Total phosphorus
TSI:	Trophic State Index
UoE:	University of Eldoret
USA:	United States of America
USAID:	The United States Agency for International Development

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CHAPTER ONE

INTRODUCTION

1.1. Background

Aquatic ecosystems such as lakes, reservoirs and rivers are essential water sources for many rural and urban communities worldwide. Humans benefit from aquatic ecosystems (lakes, reservoirs, and rivers) in several ways, including access to fish, water for agriculture and household needs, and a variety of other resources (Low *et al*., 2016; Merga *et al*., 2020). Yet, these ecosystems are experiencing water quality challenges that threaten their ecological integrity and provision of ecosystem goods and services (Bogardi *et al*., 2020; Turton, 2020).

Enrichment of water with nutrients like nitrogen and phosphorus is an example of an anthropogenic process that degrades water quality (Bashir *et al*., 2020; Bogardi *et al*., 2020). These nutrients are derived from uplands and catchment areas because of human activities such as unsustainable land use, untreated municipal and industrial waste discharge, and heavy agrochemical use (Nirmala *et al*., 1991; Sheela *et al*., 2011). As a result, these nutrients end up in lakes causing algal blooms that are attributed to shallow lakes and reservoirs turbidity (Ndungu *et al*., 2015). Lake ecosystems are particularly vulnerable, and nutrient accumulation can have long-lasting effects on their biological structure, ultimately impacting the ecosystem services provided to riparian communities (Monicayo-Estrada *et al*., 2012;

Muoria *et al*., 2015; Namugize *et al*., 2018). Therefore, there is an immediate need to protect the ecological integrity of these crucial aquatic ecosystems.

Lakes are susceptible to change over time because of external factors like their watersheds, groundwater, and climate (Garn *et al*., 2003). However, the negative effects on lakes can be mitigated through human intervention if the underlying causes are understood (Cardoso *et al*., 2009; Adejumoke *et al*., 2018). Nevertheless, the interdependencies between water and sediment chemicals, aquatic organism populations, water temperature, the bathymetry of the water body, and the nature of the catchment area make it difficult to fully understand how aquatic systems function (Melack, 1997).

Water quality in lakes varies both spatially and temporally as a result of a variety of internal and external factors including inflows, outflows, meteorological, and biogeochemical and physical processes (Ellah, 2020). Inflows introduce dissolved and particulate materials from the catchment into the lake, while outflows export materials out of the lake (Kalff, 2002). In contrast, evaporation influences the thermal conditions in the water column and concentrates solutes, while precipitation can either dilute water or introduce new solutes (Lesack & Melack, 1991; Talling, 2001). The influence of these processes varies among lakes based on climate, size, morphometry, catchment size, altitude, and latitude (Wetzel, 2001). In turn, these factors influence pH, temperature, electrical conductivity, turbidity, dissolved oxygen concentration, suspended solids, nutrients, biological oxygen demand, Chl-*a*, and lake communities (plankton and fish populations) (Wetzel, 2001; Oladipo and Williams, 2003; Walker *et al*., 2007; Ngatia *et al*., 2019).

The influence of water quality (physical, chemical, and biological) variables on the aquatic ecosystems structure and functioning have been previously reported (Svobodová *et al*., 1993; Chapman & Kimstach, 1996; Lam & Schertzer, 1999; Doyle & Smart, 2001; Oladipo & Williams, 2003; Walker *et al*., 2007; O'Gorman *et al*., 2016; Ngatia *et al*., 2019). The lake's productivity is driven by water quality variables, particularly Secchi depth, nutrients (phosphorus and nitrogen), and Chl-*a*, which in turn influence the lake's trophic status (Ellah, 2020). Therefore, trophic status of a lake (TSI) is a function of these water quality variables. Lakes and reservoirs can be oligotrophic (having poor nutrition), mesotrophic (having average nutrition), eutrophic (having plenty of nutrients), or hypereutrophic (having too many) (over-nourished) (Carlson, 1977; Brown & Simpson, 2001). Chlorophyll-*a* is one of the water quality variables used to measure the TSI of the lake (Carlson, 1977). On the other hand, chlorophyll-*a* can be used as a proxy for phytoplankton biomass, which controls the abundance of zooplankton and other primary consumers in aquatic ecosystems (Maberly *et al*., 2020). In aquatic food webs, plankton (both phytoplankton and zooplankton) are low trophic level organisms that serve as a link between primary producers and consumers (Sindt & Wolf, 2021).

The availability of nutrients and organic matter contributed by influent rivers in lakes determines the diversity and abundance patterns of phytoplankton and zooplankton (Tundisi *et al*., 2008). Environmental factors including light, temperature, the amount of organic matter, nutrients, conductivity, grazing, and water dynamics influence phytoplankton production (Tian *et al*., 2013; Jiang *et al*., 2014; Wang *et al*., 2015; Hardikar *et al*., 2017). Similarly, changes in environmental conditions dictate the community structure, abundance, and behaviour of zooplankton in lakes (Jose *et al*., 2015; Abdul *et al*., 2016). Therefore, there are trophic linkages between phytoplankton and consumer organism populations like invertebrates, zooplankton, and fish.

Understanding the spatial and temporal variation of plankton communities and how it relates to physical, chemical, and biological variables is crucial for management. For instance, shifts in primary production at the bottom of food webs have effects on higher trophic levels, which in turn modify ecosystem structure and function (Galiana *et al*., 2021). Usually, insufficient primary production is associated with poor fish recruitment and subsequent poor yields due to a lack of food (Friedland *et al*., 2012). Thus, a fishery's success or failure largely depends on phytoplankton and zooplankton communities (Abdul *et al*., 2016; Kamboj & Kamboj, 2020). Likewise, the dominance of certain groups of phytoplankton, such as blue-green algae (cyanobacteria), results in stunted fish growth and poor fish production since they are unpalatable to zooplankton (de Bernardi & Guisanni, 1990) and fish (Opiyo & Dadzie, 1994). Besides, some cyanobacteria are toxic affecting water use and the health of fish (Simiyu *et al*., 2018). Nevertheless, fishery resources are also experiencing unprecedented high fishing pressure due to increasing demand in Kenya (Yongo *et al*., 2021).

Meanwhile, wetland and shallow lake areas are being lost around the world (Finlayson & Moser, 1992). Changes in land use, such as agriculture, urbanization, and other forms of human development, have had a devastating effect on many of Africa's natural wetland lakes (Finlayson *et al*., 1992; Green *et al*., 2002; Ramdani *et al*., 2001). Lake Kanyaboli, a wetland lake in Western Kenya's Yala Wetland, faces similar problems including water quality challenges and increased fishing pressure (Aloo, 2003; Abila, 2005; Muoria *et al*., 2015). Lake Kanyaboli's catchment area, Yala Wetland, has undergone transformations like the reclamation of a fraction of the wetland for agricultural use (Muoria *et al*., 2015). The agricultural potential of the wetland was first explored by Sir Alexander Gibbs and his partners in 1954 (Okemwa, 1981). Their study findings lead to reclamation of 2,300 ha of the wetland between the 1960s and 1970 (Owiyo *et al*., 2014). Reclamation of Yala Wetland necessitated the construction of an artificial dyke across Lake Kanyaboli, effectively cutting off the lake from the wetland. As a result, in 1969, an influent canal and an outlet were built to allow water to flow into the lake from the Yala River and leave the lake, respectively. Yala Wetland and Lake Kanyaboli have since experienced human-induced changes that threaten their ecological integrity (Aloo, 2003; Abila, 2005). Agricultural activities in the catchment of Lake Kanyaboli introduce agrochemicals and nutrients (nitrates and phosphate) in the surface runoff into the lake.

Inputs of nutrients from the catchment are common because lakes are often sinks for nutrients in the landscape (Verburg *et al*., 2018). Although natural processes introduce nutrients in aquatic ecosystems, their occurrence in lakes and other aquatic environments has recently been associated with anthropogenic sources (Ngatia *et al*., 2019). Primary production in lakes relies on nitrogen and phosphorus, both of which are primarily supplied by atmospheric deposition and runoff from the catchment and surrounding areas (Verburg *et al*., 2018). Thus, the catchment area to lake ratio is important because a high ratio leads to more nutrient enrichment vulnerability (Walker *et al*., 2007). For example, Lake Kanyaboli drains a big catchment area (175km2, Odero & Odenyo, 2021), making it more vulnerable to nutrient enrichment and water quality challenges.

Furthermore, the lake is increasingly under pressure from human population growth, ensuing anthropogenic activities, and intensifying agro-industrial development efforts around the lake and the Yala Wetland (Owiyo *et al*., 2014). The Yala Wetland's dominant vegetation, papyrus (*Cyperus papyrus*) and the reed (*Phragmites australis*), which are used in handcrafts and as animal feed, are dwindling. Papyrus also provides refugium for fish in Lake Kanyaboli, in the littoral zone or through floating islands formed by these macrophytes.

Lake Kanyaboli support small-scale fisheries that thrive on the exploitation of tilapia, haplochromines, *Clarius* spp. and *Protopterus aethiopicus* (Abila, 2005). However, the lake has been experiencing increased fishing efforts derailing management and conservation efforts (Aloo, 2003; Abila, 2005). Moreover, the limnology of the lake influenced by anthropogenic activities also affect the fishery. Nevertheless, Lake Kanyaboli fishery resources are important for riparian communities. Yet temporal changes in fish composition, landing and fishing effort are poorly understood.

Lake Kanyaboli and Yala Wetland in Kenya must be protected and managed to ensure the continued supply of extractive and non-extractives goods and services. The lake and wetland provide a unique environment endowed with a rich ecosystem that supports a diversity of flora, fauna, and Lake Victoria's remnant fish species (Muoria *et al*., 2015). This has resulted in efforts to list the Yala wetland as a Wetland of International Importance under the Ramsar Convention by environmentalists and government agencies such as the Kenya Wildlife Service to ensure the conservation of its unique biodiversity and invaluable ecosystem services (Muoria *et al*., 2015). The numerous anthropogenic factors the wetland and Lake Kanyaboli are experiencing necessitated the present comprehensive study to reveal its status, which is very important to inform future interventions to protect the ecological integrity of the lake and wetland.

1.2. Statement of the problem

Most countries in Sub-Sahara Africa depend heavily on exploiting their natural resources for livelihood. These natural resources are found in impoverished rural communities where residents entirely rely on them for livelihoods (Owiyo *et al*., 2014). For example, Yala Wetland in western Kenya and its peripheral lakes (Kanyaboli, Nyamboyo, and Sare) support riparian communities' livelihoods through extractive and non-extractive services. Sustainable management of the lake and wetland is important to ensure the continued provision of ecosystem services as well as securing their ecological integrity.

Lake Kanyaboli is part of the Yala Wetland and is important for fisheries and biodiversity conservation in western Kenya (Aloo, 2003). However, extractive activities like fishing and water abstraction for irrigation from the lake and influent river can potentially change its health and trophic structure (Muoria *et al*., 2015; USAID, 2016). For example, overfishing due to increased fishing effort and the use of illegal fishing gear are common in Lake Kanyaboli (Abila, 2005). County Government of Siaya (CGS, 2015) noted that currently, Victoria tilapia (*Oreochromis variabilis*), Blue Victoria mouthbrooder (*Haplochromis nubilus*), lungfish (*Protopterus aethiopicus),* Fischer's Victoria squeaker (*Synodontis afrofischeri*), Nile tilapia (*O. niloticus*), Ripon barbel (*Labeobarbus altianalis*), and East African red-finned barb (*Enteromius apleurogramma*) populations of which once flourished in lakes Kanyaboli, Namboyo and Sare are scarce due to fishing pressure.

Similarly, Lake Kanyaboli's ecology has been negatively impacted by unsustainable agricultural activities and vegetation clearance around the lake and Yala Wetland (Abila, 2005). The use of fertilizer in the adjoining areas, livestock grazing and deposition of nutrients and organic matter from defecation and excretion of livestock pose eutrophication and ecosystem health challenges in Lake Kanyaboli (Aloo, 2003; Abila, 2005; Muvundja *et al*., 2009; Khatri & Tyagi, 2015; Muoria *et al*., 2015; Iteba *et al*., 2021). Furthermore, high electrical conductivity and/or salinity also affects the ecology of the lake and availability of portable water for people and their livestock (LBDA, 1988; Fowe *et al*., 2015; Low *et al*., 2016; Goshu *et al*., 2017; Hamid *et al*., 2020). The increased anthropogenic activities in Yala wetland and Lake Kanyaboli necessitated the gazettement of Lake Kanyaboli as a National

Reserve through legal notice No 158 of 2010 to ensure their conservation (USAID, 2016). For sustainability's sake, gazettement must be accompanied by ecological studies to inform relevant interventions and awareness creation among riparian communities.

Despite efforts to manage and conserve the wetland and its three lakes, over 2000 ha of Yala Wetland have been reclaimed for agricultural use. The wetland has faced land use changes since 1954 (Kinaro, 2008), from extensive agriculture and harvesting of wetland resources such as trees and macrophytes for fuelwood and handcrafts by riparian communities. The agricultural activities range from subsistence farming by communities to large-scale agricultural development in the reclaimed part of the wetland, such as projects implemented by the Dominion Group of Companies in 2004 (Kinaro, 2008). Poor farming practices and mechanized agriculture have contributed to suspended solids and agrochemical (herbicides and pesticide pollution) water pollution (Owiyo et al., 2014).

Lake Kanyaboli's large catchment continues to experience anthropogenic effects due to population growth that uses the Yala Wetland for livelihood activities (Abila 2008). Land use such as clearing of papyrus vegetation potentially increasing soil erosion and subsequent suspended solids transportation to Lake Kanyaboli (Abila, 2005). The visibly brownish water in the inflowing river after the rains shows signs of erosion in the lake's catchment. Likewise, the lake water retains a brownish colour during the rainy season, indicating increased levels of turbidity from suspended sediments. The highlighted interacting effects act as sources of nutrient fluxes that threaten eutrophication in Lake Kanyaboli. Similar human activities have caused increased nutrient inputs and eutrophication in Lake Victoria (Hecky, 1993; Yongo *et al*., 2021).

Lake Kanyaboli and Yala Wetland have experienced several human activities summarized chronologically in Figure 1. According to Jarosiewicz & Witek (2014) agriculture is one of the major non-point sources of nutrient loading in lakes. Thus, agricultural activities in the catchment of Lake Kanyaboli threaten the lake. In addition, Lake Kanyaboli also suffered from a lack of inflow between the 1980s and 2005 due to blockage of an artificial influent canal constructed during wetland reclamation (Mavuti, 1992; Aloo, 2003). These anthropogenic activities likely influence the lake's ecology affecting its communities (plankton and fish) and goods and services (Okemwa, 1981; Anyona, 1997; Aloo, 2003; Wilfred *et al*., 2005; Kinaro, 2008).

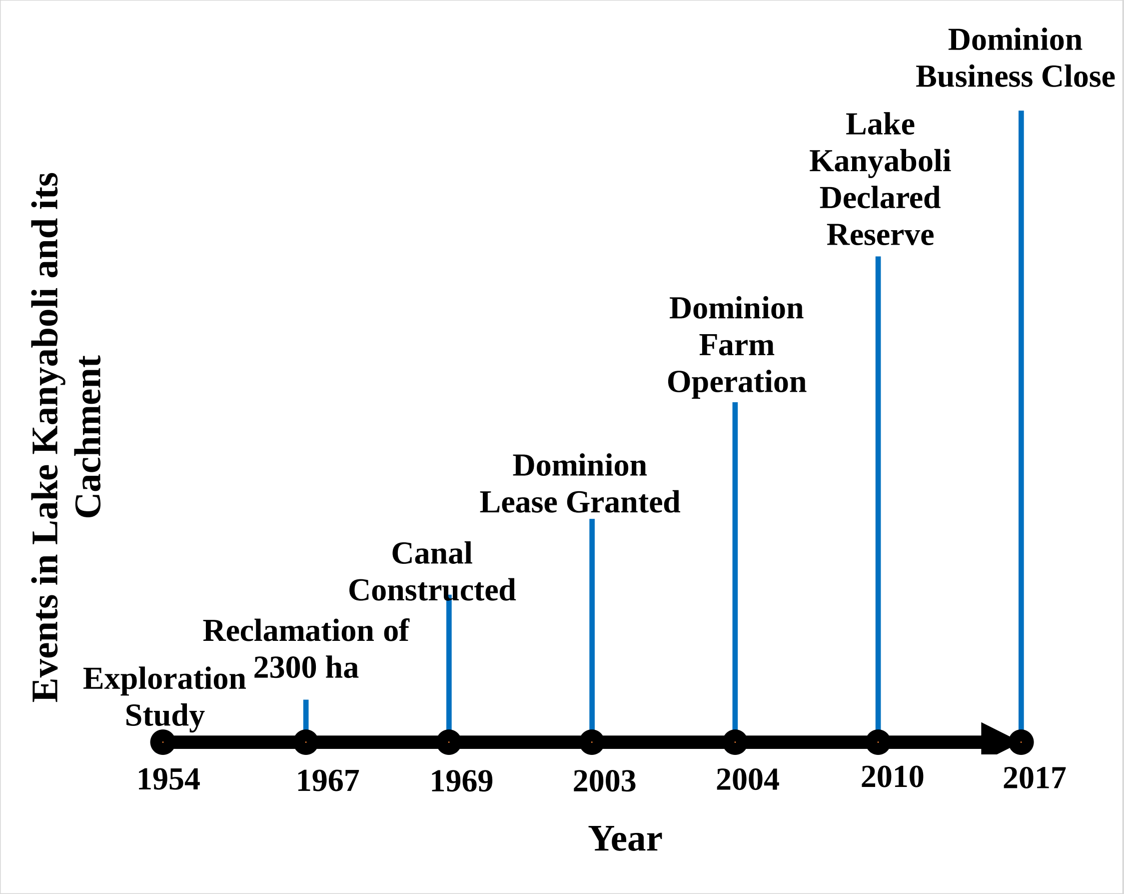

Figure 1: Chronology of events in Lake Kanyaboli and Yala Wetland

Additionally, limiting nutrients (phosphorus and nitrogen) derived from atmospheric deposition and influent loads from the landscape contribute to lake's eutrophication (Scott & McCarthy, 2010; Ngatia *et al.*, 2019). The concentration of phosphorus (P) and nitrogen (N) in the biomass at the food web base regulates the processes and functioning of the ecosystem (Sterner *et al.*, 1997). Therefore, anthropogenic activities that increase the nutrient loading from the drainage basin accelerate the eutrophication process and trophic status maturity of the lake or reservoir. For instance, land use and land cover changes due to population growth have been linked to changes in water quality variables, aquatic community structure, and inland water functioning due to nutrient enrichment from the catchment (Catherine *et al*., 2013; Kibena *et al*., 2014; Bucak *et al*., 2018; Leech *et al*., 2018; Namugize *et al*., 2018). Thus, monitoring the sources of nutrients and maintenance of nutrient balance in inland waters such as lakes is crucial. However, ecological disturbances in Lake Kanyaboli and Yala Wetland have proliferated without meaningful detailed studies to evaluate the potential effects of the interferences on lake's water quality and communities. This threatens the biological diversity of the lake because some species are likely disappearing before they are discovered (Aloo, 2003).

Therefore, this study assessed the temporal variation in physical, chemical, and biological variables and communities (plankton and fish) in Lake Kanyaboli from 1981 to 2020. These data are crucial in developing policies and strategies for conserving and managing the lake and similar waterbodies to ensure sustainable provision of ecosystem goods and services to the people.

1.3. Justification of the study

Understanding the variability in physico-chemistry and biology of inland waters (e.g., lakes and reservoirs) and linking them to drivers of water quality and food web dynamics is important for developing strategies for their management and conservation (Pinto-Coelho *et al*., 2005; George *et al*., 2012; Thakur *et al*., 2013; Hemraj *et al*., 2017). The study of water quality (physical, chemical, and biological) characteristics of Lake Kanyaboli is important because these can strongly be influenced by human activities on spatial and temporal scales.

Monitoring studies are essential because the impact of human influence on inland waters such as Lake Kanyaboli is expected to increase in the future due to rapid increase in human population and livestock pressure in the catchment area (Yala Wetland) (Mwaura *et al*., n.d).

The Yala Wetland is under agriculture, and various crops are grown with varying fertilizer requirements (Wilfred *et al*., 2005), threatening Lake Kanyaboli water quality. The fertilizers contain nutrients such as nitrogen and phosphorus, which would make the lake eutrophic if unchecked. This would compromise the survival of the various organisms in the lake, including plankton and fish; therefore, such interaction needs to be understood. The lake and its catchment provide significant goods and services necessary for the ecosystem's stability and livelihoods to local communities. Recent agro-industrial developments within Yala Wetland are further expected to exacerbate the situation hence the need to monitor water quality challenges in the lake and their effect on biological communities.

The Yala Wetland is a significant part of Lake Kanyaboli and harbours the critically endangered sitatunga antelope (*Tragecephalus spekeii*). The wetland is one of the sixty Kenyan bird areas, with at least 172 bird species, some of which are biome-restricted or threatened (Muoria *et al*., 2015; CGS, 2015). These make the area a potential tourist destination for bird watching and water sports in Lake Kanyaboli. Furthermore, the lake harbours Lake Victoria's remnant fish species that were impacted by Nile Perch (*Lates niloticus*) introduction (Aloo,2003; Abila, 2005; Owiyo *et al*., 2014). This ringfences the ecological and conservation importance of the Yala Wetland; hence the present study directly responds to the need for physical, chemical and biological scientific data for policy development and decision-making purposes. The study further contributes to our understanding of small water bodies ecology.

Kenya is obligated to ensure the wise use of Yala Wetland regardless of its Ramsar status because the country is a signatory of the Ramsar Convention. However, the continued agricultural mechanization in Yala Wetland affects the wetland and small lakes within the wetland, such as Lake Kanyaboli, Namboyo and Sare. These anthropogenic effects need scientific documentation to ensure evidence-based management. Sustainable management of Lake Kanyaboli ecosystem requires data on water quality, nutrient dynamics, fish species composition, catch and effort, plankton community structure, and phytoplankton steady-state. Conservation of the lake and its catchment will help preserve the biodiversity and safeguard the livelihood of riparian communities that rely on exploitation of these natural resources.

1.4. Study objectives

1.4.1. General objective

The general objective of this study was to assess the spatio-temporal variation in trophic status, water and nutrient budget and communities of Lake Kanyaboli to inform lake management and conservation of its biodiversity.

1.4.2. Specific objectives

i. To determine historical changes in the limnological characteristics (DO, temperature, EC, Chl-*a*, NH_4^+, NO_3^-, NO_2^-, SRP, TP, TN) and trophic status of Lake Kanyaboli from 1981 to 2020.

ii. To determine factors that influence nutrient (mainly nitrogen and phosphorus) dynamics in Lake Kanyaboli using a mass balance model.

iii. To assess spatio-temporal dynamics in environmental conditions and how they drive plankton (zooplankton and phytoplankton) communities in Lake Kanyaboli.

iv. To determine temporal changes in fish species composition, catch and fishing effort in Lake Kanyaboli.

1.5. Study hypotheses

H_{01}: There are no historical changes in the limnological characteristics (DO, temperature, Chl-*a*, EC, NH_4^+, NO_3^-, NO_2^-, SRP, TP, TN) and trophic status of Lake Kanyaboli.

H_{02}: Temporal nutrient inputs and outputs variations do not affect nitrogen and phosphorus budgets in Lake Kanyaboli.

H_{03}: Variation in physico-chemical variables does not drive annual plankton community variation in Lake Kanyaboli.

H_{04}: There are no temporal changes in fish species composition, catch and fishing effort in Lake Kanyaboli.

CHAPTER TWO

LITERATURE REVIEW

2.1. Limnological characteristics of lakes

Understanding the functional dynamics of lentic ecosystems is multifaceted because of the interdependencies among chemicals in the sediments, aquatic organism populations, water physico-chemistry, the water body morphometry, and the nature of the drainage basin (Chapman & Kimstach, 1996; Melack, 1997; Kalff, 2002; O'Gorman *et al*., 2016; Doubek & Carey, 2017). These aquatic ecosystems are even more challenging to understand when one considers the humans part of the dynamics of these systems (Melack, 1997; Walker *et al*., 2007; Xu *et al*., 2019; Paerl *et al*., 2015). Concerns over the impact of anthropogenic activities on rivers, ponds, and lakes continue to grow. This is due to the negative effect of anthropogenic activities such as sewage from cities, effluents from factories, agrochemicals from various agricultural operations, and pathogens that accumulate in these water bodies (Nirmala *et al*., 1991). Furthermore, development activities tend to clear forests and wetlands, negatively affecting rivers and lakes' status. Understanding the limnological dynamics of lentic ecosystems is important as they form critical life support systems for biodiversity and riparian communities through the provision of diverse ecosystem services.

Nirmala *et al*. (1991) reported that Pookot Lake of Wayanad district in India was once a serene freshwater lake before the negative effect of anthropogenic activities such as extensive deforestation and agricultural activities. These result in alterations in nutrient cycles, pollution, temperature, stratification dynamics, food web structure, and the abundance and phenology of ecologically or socio-economically important organisms (Hampton *et al*. 2018). Previous studies in Lake Kanyaboli have also reported changes in the physico-chemical variables due to anthropogenic activities (such as agriculture) in the reclaimed Yala Wetland (Okemwa, 1981; Anyona, 1997; Aloo, 2003; Wilfred *et al*., 2005; Kinaro, 2008).

Physical, chemical, and biological variables, mainly Secchi depth, nutrients and chlorophyll-*a*, affect the lake's productivity, which afterwards affects the trophic status of the lake. The various water quality variables that influence the ecology of lentic waters are described in detail below.

2.1.1. Dissolved oxygen

Dissolved oxygen (DO) is one of the major water quality variables for the survival of all aquatic organisms. The physical, chemical, and biological processes that produce or consume oxygen determine dissolved oxygen levels in a water body (Bengtsson *et al*., 2012). Dissolved oxygen is commonly considered an indirect indicator of potential primary production and is essential for aquatic organisms' aerobic biochemistry metabolism (Ergönül *et al*., 2020). The concentration of DO below 5 mg L^{-1} may negatively affect the functions and survival of biological communities, and concentrations below 2 mg L^{-1} are considered hypoxic and can lead to the death of most fishes (Chapman & Kimstach, 1996).

The oxygen content in lakes is influenced by various variables and processes such as salinity, temperature, turbulence (wind action or upwelling), the photosynthetic activity of aquatic plants, and atmospheric pressure (Bengtsson *et al*., 2012). For example, an increase in temperature and salinity results in a decrease in oxygen solubility. The solubility of oxygen in freshwater ranges from 15 mg L^{-1} at 0° C to 8 mg l^{-1} at 25° C (Bartram & Balance, 1996). Wind action is also considered one of the essential factors in dissolved oxygen availability due to the contribution toward the physical re-aeration of shallow lake surface water (Lindenschmidt and Chorus, 1997; Kirke and el Gezawy, 1997). Dissolved oxygen shows diel and seasonal variations in response to water temperature and biological activity changes (photosynthesis and respiration) (Bartram & Balance, 1996). Oxygen is sometimes available only during the daytime (D’Autilia *et al*., 2004).

Biological processes such as respiration (including decomposition) are DO dependent hence they reduce DO concentration. Thus, waste discharges (high in nutrients and organic matter) potentially decrease DO concentrations attributable to increased microbial activity (USEPA, 2008). This may lead to anaerobic conditions in severe cases, particularly at the lake bottom due to decaying, sinking materials. The assessment and monitoring of DO is important because of their involvement or influence on almost all chemical and biological processes in lakes and other water bodies (USEPA, 2008).

2.1.2. Water transparency

The transparency, known as a Secchi depth (SD) measurement, is the distance the light penetrates a water body. It is a valuable parameter in estimating the euphotic lake zone and the trophic state of the ecosystem and gives an idea of the aquatic life condition for organisms (Istvánovics, 2010). According to Bhatnagar *et al.* (2004), water clarity ranges from 30-80 cm for the good condition of fish, and a Secchi depth level of less than 12 cm causes stress. The Secchi depth (transparency) between 30 and 40 cm indicates the optimum productivity of an ecosystem, and it is the recommended level for optimum fish growth in an open water body (Santhosh & Singh, 2007).

Shallow lakes are more susceptible to water clarity because of nutrient loads and the suspended solids from the catchment (Melack, 1997). Unlike deep and voluminous lakes that are buffered by the lake volume, shallow lakes like Lake Kanyaboli constantly suffer from sediment inputs that affect water clarity (Okemwa, 1981; Anyona, 1997). The problem is exacerbated by clearing vegetation and reclamation of wetland, which helps filtrate excess nutrients before reaching the lake to reduce nutrient enrichment and sedimentation (Muoria *et al*., 2015).

2.1.3. Water temperature

Temperature is critical in determining the environment's suitability to support life because it affects other water components, such as saturation and concentration of dissolved oxygen, which affects the survival of plankton, fish, and other organisms (Bartram & Balance, 1996).

Additionally, temperature helps regulate feeding, metabolic activities, and egg development in fish and zooplankton (O'Gorman *et al*., 2016). Seasonal changes directly affect water temperature through induced fluctuations based on air temperature (Hren & Sheldon, 2012). The surface layer of lakes usually undergoes daily temperature variations while the lake's bottom remains constantly cold (Jørgensen, 1980).

Water bodies undergo continuous temperature changes and normal climatic variations (Bartram & Balance, 1996). These fluctuations occur in the short-term (diel) and longer-term (seasonally). Water temperature is influenced by time of day, season, latitude, altitude, air circulation, cloud cover, and the flow and depth of the water body, with the impact being more pronounced in shallow (Chapman & Kimstach, 1996). On the other hand, temperature influences ecological processes in ecosystems and, therefore, the concentration of many variables. The increase in water temperature leads to a rise in the rate of chemical reactions and the rate of the evaporation and volatilization of substances from the water (O’Gorman *et al*., 2016). The decrease in the solubility of gases in water, such as O_2, CO_2, N_2, CH_4, and others, is also attributed to the increased temperature (Bengtsson *et al*., 2012).

The metabolic rate of aquatic organisms is also influenced by temperature; hence in warm waters, respiration rates increase, leading to increased oxygen consumption and decomposition of organic matter (O’Gorman *et al*., 2016). The growth rates of phytoplankton are also influenced by water temperature, such that the proliferation of phytoplankton is common in warm waters and is rich in nutrients, leading to algal blooms and associated water turbidity (Bartram & Balance, 1996). Temperature fluctuation is maximal during the wet season and less pronounced during the wet season, particularly in shallow waters (Chapman & Kimstach, 1996).

Temperature also affects the mixing of different layers of the stratified lake. Although tropical shallow lakes and reservoirs receive high solar radiation, which is important in setting local stratification, cold inflow, differential heating, and reservoir releases affect stratified layers' formation, as observed in Kranji Reservoir in Singapore (Xing *et al*., 2014). In tropical lakes, it is observed that the thickness of the diurnal mixed layer is less stable in shallow as well as deep lakes than in temperate lakes, ascribed to higher radiative heating (Lewis, 1987; Talling & Lemoalle, 1998).

2.1.4. Potential of hydrogen (pH)

The potential of hydrogen (pH) refers to measuring the acidity or alkalinity of water. The pH range of 6.5 - 9.0 is, for instance, suitable for maximum growth and abundance of plankton. In contrast, low pH creates stress in most organisms, leading to a decrease in metabolism and death in severe cases, hence reducing biodiversity (Oladipo & Williams, 2003). Photosynthesis increases water's pH by absorbing carbon dioxide (Moss, 1973). However, respiration has a reverse effect of decreasing pH through oxygen consumption and the release of carbon dioxide. This phenomenon results in more alkaline conditions during the day, while acidic conditions are observed at night because photosynthesis occurs during the day while respiration peaks at night (Bartram and Balance, 1996). Chapman & Kimstach (1996) reported that temperature and solar radiation accelerate photosynthesis, increasing carbon dioxide absorption and resulting in alkaline conditions due to the production of OH^- ions.

The continuous monitoring of pH changes, together with conductivity, act as an indicator of the presence or absence of certain influents in an ecosystem (Bartram and Balance, 1996). The photosynthesis and respiration cycles of algae can be responsible for diel variations of eutrophic water pH. The pH of most natural water ranges from 6.0 to 8.5, but lower values commonly occur in dilute waters, high in organic content, and higher values are observed in eutrophic waters and lakes with high salinity (Chapman & Kimstach 1996). Below a pH of 6.5, fish species and planktons show slow growth (Lloyd, 1992; Goshu *et al.*, 2017). The pH can be influenced by many natural factors therefore, it should be determined *in situ* or immediately after the collection of the sample (Bartram & Balance, 1996).

2.1.5. Electrical conductivity

Electrical conductivity (EC) is sensitive to variations in dissolved solids and temperature (Chapman & Kimstach, 1996). In most freshwater, EC ranges between 10 and 1,000 µS $cm^{-1,}$ although it may exceed 1,000 µS cm^{-1} in polluted waters or those receiving large quantities of influents from agricultural land (Bartram & Balance, 1996). This characteristic makes EC an alternative indicator of mineral content when other methods cannot be used easily; hence it is measured to establish pollution zones such as an effluent discharge or the influence of influent water (Bartram & Balance, 1996). Electrical conductivity is usually measured *in situ* using a multimeter probe.

2.16. Chlorophyll-*a*

The Chl-*a* is routinely quantified using standard procedures based on extraction from disintegrated cells in an organic solvent such as methanol, ethanol, or acetone, before determination by spectrophotometry (Parsons & Strickland, 1963; Lorenzen, 1967), fluorometry (Holm-Hansen *et al*., 1965), or chromatography (Jacobsen, 1978; Otsuki & Takamura, 1988). The process starts with filtration separating algal cells from the water. This is done using glass-fibre filters because of the large filtration capacity and flow rate (Aminot & Rey, 2001). Whatman GF/C and GF/F glass-fiber filters are used by most researchers with varying pore sizes (0.7 or 0.45 lm) and diameters (25 or 47 mm) (Qin *et al*., 2013). However, these methods are time-consuming and require a standard sampling procedure and transport to a laboratory as well a large volume of samples (Gregor & Maršálek, 2004). These processes are not immune to the introduction of variability and errors.

One vital Chl-*a* characteristic is its fluorescence which revolutionalized the *in vivo* determination of Chl-*a* (Gregor & B. Maršálek, 2004). The *in situ* fluorometers have been developed and are installed with probes capable of detecting eukaryotic algae and phycocyanin of cyanobacteria (Desiderio *et al*., 1997; Asai *et al*., 2001; Beutler *et al*., 2002; Leboulanger *et al*., 2002). *In situ* and continuous measurement of Chl-*a* is advantageous over discrete sampling because some phytoplankton species, such as cyanobacteria, can be concentrated in a very thin layer (Gregor & Maršálek, 2004). Nevertheless, field-based fluorometers are expensive; hence laboratory-based methods are commonly used.

Eutrophication is a major important water quality problem affecting lakes and reservoirs, caused by multiple physico-chemical factors that lead to algal blooms. Shallow lakes, ponds and ditches are more vulnerable to eutrophication (Janse, 2005). As a representative eutrophication index, the concentration of Chl-*a* is a key indicator monitored by environmental managers (Li *et al*., 2017). The most influential factors on Chl-*a* may depend on the different water quality patterns in lakes. Photosynthetic phytoplankton contains chlorophyll pigment that is used to estimate biomass and primary productivity in aquatic systems (Arrigo *et al*., 2008) such as lakes. The Chl-*a* concentration varies spatially and temporally. Seasonal variation of Chl-*a* concentration in lakes is linked to a load of nutrients (phosphorus and nitrogen) (Dimberg, 2011). Excessive nitrogen and phosphorus loads are responsible for the lake's shift from oligotrophic to hypertrophic conditions (Paerl *et al*., 2015), leading to increases in harmful cyanobacteria blooms, threatening lake ecosystems (Xu *et al*., 2014). Nevertheless, Chl-*a* data in most lakes especially developing such as Lake Kanyaboli, Kenya is scarce.

2.1.7. Nutrients

An aquatic system's nutrient load (including total nitrogen, total phosphorus, nitrate, nitrite, ammonium and soluble reactive phosphorus) is either from internal (autochthonous) or external (allochthonous) sources affected by inflows and outflows. Nutrients are elements living organisms use as nourishment (Walker *et al*., 2007). Some nutrients, referred to as macronutrients, such as nitrogen (N) and phosphorus (P), are needed in relatively large supplies. In contrast, other nutrients, such as iron, zinc, and copper, are needed in a comparatively small supply (known as micronutrients) (Hecky & Kilham 1988).

Lakes and reservoirs derive nutrients from various sources, including bedrock, atmospheric deposition, vegetation and animal life in and around the water body, and input from human activities (Walker *et al*., 2007). Phosphorus and nitrogen often limit the amount of growth by primary producers, hence are termed “limiting nutrients” (Ngatia *et al*., 2019). Besides limiting primary production, phosphorus and nitrogen are also “causal variables” because they lead to the proliferation of primary producers when in excessive amounts (Walker *et al*., 2007); hence, much attention is paid to phosphorus and nitrogen when developing nutrient criteria (USEPA, 2000).

Although phosphorus and nitrogen can be sourced naturally, in recent years, their occurrence in lakes, reservoirs, rivers, and the aquatic environment has been associated with anthropogenic sources (Ngatia *et al*., 2019). Thus, many countries have implemented mechanisms to control point sources, but controlling non-point sources of these nutrients, especially agricultural ones, remains a challenge (Carpenter *et al*., 1998). Lake Kanyaboli is one of the wetland lakes experiencing human impact in terms of nutrient enrichment due to agricultural activities in its catchment area. However, the magnitude of these nutrient loads is not well understood due to limited studies; hence the current study aims to generate data necessary for understanding nutrient dynamics in the lake to help device nutrient pollution management approaches and policies.

2.1.8. Trophic status in lakes

Lakes are commonly classified according to their trophic state, which describes how green the lake is as measured by the amount of algal biomass in the water (Brown & Simpson, 2001). Physico-chemical parameters, especially chlorophyll-*a* and Secchi depth and nutrients such as phosphorus and nitrogen, are essential for determining the trophic status (Carlson, 1977). The trophic status of water bodies, such as lakes and reservoirs, are classified as oligotrophic, mesotrophic, eutrophic and hypereutrophic based on their nourishment. A lake's trophic state indirectly measures the concentration of nutrients, chlorophyll-*a*, and water clarity instead of measuring algae biomass directly (Secchi depth) (Carlson, 1977).

The calculation of the Carlson trophic status index (TSI) of a lake or any other water body is done using Secchi depth transparency (SD), chlorophyll-*a* (Chl-*a*), total phosphorus (TP) and total nitrogen (TN) equations (Carlson 1977). The TSI is a commonly used classification of surface waters' trophic conditions (Ellah, 2020). This is despite the acquisition of input data for TSI involving labour- and time-consuming processes, such as field water sample collection, transportation, sample filtration, and chemical analysis (Dodds, 2007). Furthermore, the labour intensiveness of data acquisition inhibits the rapid assessment and characterization of TSI in inland waters undergoing a rapidly changing environment (Shi *et al*., 2019).

The lake's trophic state is a function of nutrient availability that supports primary production, affecting the clarity of the lake and estimated Chl-*a*. High nutrient levels in the lake lead to eutrophication, i.e. directional movement of the lake's state from the lower trophic state to the eutrophic state over time (Brown & Simpson, 2001). Natural and human factors cause the eutrophication of lakes. Natural eutrophication is the process by which lakes gradually become aged and more productive; however, this process is accelerated by excess nutrients (primarily N and P) from human activities disposed into lakes, mainly as untreated or partially treated domestic sewage, runoff from agricultural fields, industrial input etc. (Sheela *et al*., 2011). Phosphorus is considered an essential nutrient for the control of eutrophication. The algae's excessive growth or 'blooms', promoted by phosphates, changes water quality. As the overcrowded plants die off, the decaying vegetation depletes the lake's oxygen supply, leading to the death of aquatic organisms such as fish (Sheela *et al*., 2010).

There is a relationship between TSI indices which explains the dynamics of nutrients and primary production in the lake. Havens (2000) noted that the agreement between values of TSI (Chl-*a*) and TSI (SD) infers that algae dominate light attenuation. On the other hand, the lower TSI (Chl-*a*) than TSI (TP) indicates less algal material present than expected based on TP. Therefore, these relationships are utilized to assess nutrient accumulation in the aquatic system, and small shallow lakes are ideal for assessing such relationships. Nutrients that affect a lake’s trophic status are affected by drought events and the associated period of low water levels that lead to elevated temperature, conductivity and chlorophyll-*a* levels in these shallow lakes, which probably led to high photosynthetic rates resulting in increased dissolved oxygen and pH and decrease in total nitrogen concentration (Sharip *et al*., 2019).

The TSI (Chl-*a*) is considered the definitive indicator of trophic status because Chl-*a* is a direct measure of algal biomass (Bolgrien *et al*., 2009). The trophic status index ranges from ultra-oligotrophic (approximately TSI = 0) to hyper-eutrophic (TSI > 100). USEPA (1998) noted that lakes with TSI > 50 often have taste or odour problems, nuisance macrophytes, algal blooms and hypolimnetic anoxia. These TSI indices are used for management purposes; for instance, South Dakota lakes for recreation, fish propagation, and aesthetics are considered impaired when TSI exceeds 55.5 (Bolgrien *et al*., 2009). Despite such management benefits of these indices, literature on their use in many tropical lakes, such as Lake Kanyaboli, is lacking and non-existent. Therefore, the use of the TSI index in the present study to assess the trophic state of Lake Kanyaboli is important for the lake and similar lakes in the region and elsewhere. This study aims to help enrich the literature on the drivers of the water quality in Lake Kanyaboli.

2.2. Nutrient and water mass balance in lakes

The nutrient balance refers to the difference between the nutrient inputs entering a system and the nutrient outputs leaving the system (Verburg et al., 2018). Nutrient balance provides data on a range of lakes, including efficient sinks of phosphorus and the ones that have almost continuous phosphorus release from anaerobic (Laugaste, 1994) or aerobic sediments (Lofgren, 1987). Nutrient balance analysis can be a valuable mechanism for measuring intimate processes like denitrification when combined with sediment investigations (Dudel & Kohl, 1991; Jensen *et al*., 1994; Ahlgren *et al*., 1994).

The productivity in aquatic systems is sustained by nutrient availability. Primary production in lentic water bodies such as lakes is mainly controlled by nutrient concentration in the euphotic zone (Ngatia *et al*., 2019). Nutrient input comes from external or internal sources (Wetzel, 2001). Water enters lakes from precipitation, and runoff from the surrounding land, streams, and groundwater; these waters bring nutrients to the lake (Kalff, 2002). Runoff that enters lakes first passes through the surrounding wetlands, slowing the water velocity and removing sediments, nutrients, and pollutants (Ryken *et al*., 2015). Therefore, wetlands are considered important filters that regulate the nutrient balance of lakes. For example, Yala Wetland, a catchment for Lake Kanyaboli, is dominated by *Cyperus papyrus* which helps filter nutrients (Kelderman *et al*., 2007) that could have otherwise found their way to the lake. Ryken *et al*. (2015) noted that wetlands function as a sediment trap and buffer peak discharges, but many of the papyrus wetlands (such as Yala Wetland) in the Lake Victoria basin are under massive pressure from human settlement and agricultural activities. These threaten nutrient enrichment in Lake Kanyaboli, which may also influence biological communities.

Furthermore, Noges (2005) noted that earlier stagnation and absence of spring turnover affect nutrient dynamics, as observed in Lake Verevi, Estonia. The prolonged warm and dry conditions also increase chemical constituents' retention due to early stratification, which accumulates higher concentrations of decaying matter in the hypolimnion (Blenckner, 2001). However, unstable environmental conditions such as temperature may only last a few hours (Lewis, 1987; Talling & Lemoalle, 1998). For instance, Kranji Reservoir in Singapore is associated with short-term stratification that is susceptible to cold water inflow regime from the catchment area and reduction in temperature overnight (Xing *et al*., 2014). These events lead to mixing that resuspends nutrients hence crucial in understanding nutrient budgets/ balances in shallow lakes to manage pollution and eutrophication.

Lake Kanyaboli, a shallow tropical lake found within the Lake Victoria basin, Kenya continues to experience anthropogenic effects from the Yala Wetland catchment due to population growth that uses the Yala Wetland for livelihood activities, increasing the risk of nutrient enrichment (Abila 2008). Land use in the Lake Kanyaboli catchment is predominately agriculture, either subsistence or commercial, leading to fertilizers and agro-chemicals (Muoria *et al*., 2015). The catchment is also experiencing clearance of papyrus vegetation which increases runoff erosion threat and subsequent sedimentation in the lake (Abila, 2005). Besides sedimentation, the highlighted interacting effects act as sources of nutrient fluxes that threaten eutrophication in Lake Kanyaboli. Similar events, such as agriculture, and deforestation, have caused increased nutrient inputs and eutrophication in Lake Victoria (Hecky, 1993).

Water balance dynamics also influence the nutrient balance of the lake, with shallow lakes and reservoirs being more susceptible due to evaporation affecting water quality (Fowe *et al*., 2015) and the situation being more severe in the lentic system. The inflows and outflows affect the mass nutrients of nitrogen and phosphorus transported into and out of the lake. The inflowing rivers carry nutrients loads into lakes during the rainy season, which is crucial for primary production supporting consumers such as zooplankton, which are essential food items for the fish community (Wondie *et al*., 2007). However, these unchecked nutrients lead to lakes' eutrophication, affecting aquatic life and water supply to riparian communities. Therefore, it is important to determine nutrient loads in Lake Kanyaboli and their fate to understand nutrient-related water quality challenges.

The lake also loses water to the atmosphere through evapotranspiration, downstream receiving water from an outlet, and groundwater through seepage from the lake bottom affecting nutrient availability (Walker *et al*., 2007). Downstream water loss from a lake results in the reduction of nutrient load (Noges, 2005). According to Walker *et al*. (2007), changes in the characteristics of lake waters occur with seasonal temperature and solar radiation fluctuations, particularly regarding the mixing of the upper and lower lake waters. Thus, besides the decrease of anthropogenic nutrient loading resulting from the applied rehabilitation measures, climate variability may have an essential role in the year-to-year differences in the nutrient balance (Noges, 2005).

Estimating nutrient mass balance in this lake is crucial for managing nutrient input and preserving these and similar fragile ecosystems. Aquatic systems play different roles in nutrient cycling, i.e., they either act as sources of nutrient exportation or sinks (when they absorb or retain most of the incoming nutrients. Lake Kanyaboli is limnologically poorly studied despite rampant anthropogenic activities in its catchment area that influence nutrient dynamics. Besides, such small lakes are susceptible to nutrient loss due to shorter water residence time which affects the primary production in these lakes (Xu *et al*., 2021).

It is important to understand the water balance dynamics in the lake due to the social and economic services derived from it, such as fishing, water for domestic use and livestock watering. The lake's decline or overflow potentially affects the sustainable development and livelihood of the riparian communities. Shallow lakes and reservoirs are susceptible to evaporation; hence, inflows and stable rainfall regimes are critical to maintaining good water quality (Fowe *et al*., 2015). The inflows and outflows affect the mass nutrients of nitrogen and phosphorus transported into and out of the lake. For proper lake-level management, accurate historical and current streamflow data are needed. Streamflow and outflow data are unavailable for Lake Kanyaboli; hence the present study is vital in generating baseline data to document and improve understanding of the water and nutrient balance in the lake to inform management for continued extractive and non-extractive benefits.

2.3. Phytoplankton and zooplankton communities in lakes

Understanding the structure and composition of plankton communities and their ecology requires a combination of biotic and abiotic factors due to their dynamism (Pinel-Alloul *et al*., 1999). Phytoplankton and zooplankton diversity and abundance have been utilized as water quality indicators since their composition and biomass are related to the availability of nutrients and particulate materials contributed by influent rivers (Thakur *et al*., 2013).

Phytoplankton is an assemblage of heterogeneous microscopic algal forms of aquatic systems whose movement depends on water currents (George *et al*., 2012). Their survival and proliferation depend on either one or a combination of the following factors: temperature, light (influenced by turbidity), composition and quantity of organic matter, conductivity, grazing, and nutrients (Tian *et al*., 2013; Jiang *et al*., 2014; Wang *et al*., 2015; Hardikar *et al*., 2017).

Temperature changes affect dominant species, abundance, and biomass of phytoplankton. At the same time, nutrient concentration (particularly phosphorus and nitrogen) is the limiting factor for phytoplankton growth, with high nutrient concentrations resulting in phytoplankton blooms in lakes and reservoirs (Wang *et al*., 2015). In turn, phytoplankton composition influences nutrient recycling, grazing, particle sinking, and food webs (Cetinic *et al*., 2006). The quality and quantity of phytoplankton and their seasonal patterns have been successfully utilized to assess the water quality and its capacity to sustain heterotrophic communities besides acting as water quality indicators (George *et al*., 2012). Trophic linkages exist between the phytoplankton as primary producers and populations of consumer organisms, including bacteria, zooplankton, benthic invertebrates, and fish.

Zooplankton groups in freshwater systems are predominantly represented by Cladocera, Copepoda, and Rotifera (Ismail *et al*., 2019). Zooplankton that consumes phytoplankton and other fine particles in the water column play a significant role in energy flow in aquatic food webs (Okogwu, 2010; Ismail *et al*., 2019). Zooplankton behaviour, abundance, and community structure are driven by changes in environmental conditions (Jose *et al*., 2015; Abdul *et al*., 2016). Hence, the changes in water quality can cause variations in zooplankton species assemblages (Pinto-Coelho *et al*., 2005). This can assist in the long-term monitoring of the zooplankton community structure by providing helpful information on environmental changes.

Assessment of the spatio-temporal variation in plankton communities in relation to environmental variables is vital for management decisions. For instance, changes in primary production at the base of food webs have cascading effects on other trophic levels, ultimately altering ecosystem structure and functioning (Galiana *et al*., 2020). Usually, insufficient primary production is associated with poor zooplankton community and secondary consumers, such as fish recruitment, resulting in poor yields due to a lack of food (Friedland *et al*., 2012; Abdul *et al*., 2016; Kamboj & Kamboj, 2020).

Nevertheless, how plankton communities in small water bodies maintain stability, or lack thereof, in response to changes in environmental conditions is poorly understood. The changes in plankton communities in response to environmental factors linked to seasonality and human activities have previously been documented (Sommer *et al*., 1986; Talling, 1987; Pijanowska, 1990; Chang *et al*., 2003). These fluctuations in plankton communities agree with the plankton paradox (Hutchinson, 1961), which postulates that most systems exhibit non-equilibrium conditions in community composition due to changes in physical, chemical and biological factors. Stable environmental conditions are associated with the dominance of a few competitive species leading to low diversity as per the competitive exclusion principle (Figueredo & Giani, 2009).

Although variation in environmental conditions in lakes is a function of season, this (seasonality) is not well pronounced in tropical regions due to the largely stable temperature and the potential of seasonality to influence plankton communities has not been extensively studied (Talling, 1987). Shallow lakes in tropical regions are associated with short-term thermal stratification, easily weakened by wind or convection at night (Figueredo & Giani, 2009; Yang *et al*., 2018). In contrast, phytoplankton composition in temperate regions is influenced by recurrent seasonal temperature fluctuations, stratification, light, and nutrient availability (Arhonditsis *et al*., 2003; Grover & Chrzanowski, 2006). Therefore, adaptation to changing environmental factors by phytoplankton and zooplankton is crucial to their colonization and succession.

Besides short-term studies on the diversity and composition of planktons in Lake Kanyaboli (Mutune *et al*., 2005; Wilfred *et al*., 2005; Babu *et al*., 2015), long-term studies on environmental factors as drivers of plankton assemblages are limited. The situation is common across the Afro-tropics, resulting in insufficient long-term plankton ecology data (Aka *et al*., 2000; Figueredo & Giani, 2009; Fonseca & de Mattos Bicudo, 2011). More emphasis is often placed on economic benefits studies over ecological, conservation and aesthetic benefits derived from small water bodies. For example, long-term ecology and water quality studies are available on large lakes of economic importance, including Kenya’s

Lake Naivasha (Ballot *et al*., 2009) and Baringo (Walumona *et al*., 2021), as well as the African great lakes such as Victoria (Sitoki *et al*., 2010), Kivu (Hyangya *et al*., 2021), Malawi (Macuiane *et al*., 2016), and Tanganyika (Kelly *et al*., 2017). Lately, cyanobacteria-dominated lakes have been given more emphasis due to cyanobacteria toxicity that affects domestic water supply, fish health, as well as overall ecological and economic consequences (Bicudo *et al*., 2007; Jeppensen *et al*., 2007; Simiyu *et al*., 2018; Kimambo *et al*., 2019). Therefore, long-term monitoring studies are important crucial to help understand anthropogenic activity's influence on aquatic communities and water quality for better management of small water bodies, including Lake Kanyaboli.

Studies on phytoplankton also explain the theory of equilibrium state in lakes, as Sommer et al. (1993) illustrated. The authors noted that a phytoplankton community is considered to be at equilibrium when; (i) 1, 2, or 3 species contribute to more than 80% of the biomass, (ii) their existence (or coexistence) persists for more than two weeks, and (iii) the total biomass does not increase significantly during that period. Plankton communities are seldom at an equilibrium state in small shall lakes like Lake Kanyaboli because of shorter water residence time which increase the flushing of nutrients, making them unavailable to phytoplankton, hence regulating the growth of a few competitive species (Scheffer *et al*., 1993).

Previous studies reported that cyanobacteria were dominant in Lake Kanyaboli, accounting for more than 50% of the sample, while diatoms, green algae, and euglenophytes made up the remainder (Aloo, 2003; Wilfred *et al*., 2005; Babu *et al*., 2015). Cyanobacteria dominance in the lake is related to the nutrients supply from the catchment (Wilfred *et al*., 2005). Moreover, the area has a high livestock population watering in the lake or grazing in the catchment, which is another source of nutrients from the excretes (dung and urine) (Masese *et al*., 2020a; Iteba *et al*., 2021). According to Shevah (2017), the nutrients and carbon sources from livestock are fodder for the proliferation of phytoplankton and bacteria in aquatic systems. Furthermore, eutrophic lakes are predominantly dominated by cyanobacteria because the utilize CO_2 efficiently at high pH levels, outcompeting other groups of algae (King, 1970).

The zooplankton groups belonging to copepods (*Thermocyclops* spp.), cladocerans, and rotifers (*Brachionus* spp.) have been documented in Lake Kanyaboli (Aloo, 2003; Matune *et al*., 2005). However, the lake has a poor animal plankton community linked to the dominance of less palatable cyanobacteria (Aloo, 2003; Matune *et al*., 2005). The unpalatability of cyanobacteria limits their use as food by consumers (e.g. zooplankton), which is partly why they dominate water bodies while inhibiting the proliferation of the zooplankton community (De Bernardi & Guisanni, 1990). The data generated in the present study are important because they help understand the drivers of plankton community structure and the equilibrium state of Lake Kanyaboli, Kenya, a small tropical lake. The data are crucial in developing biomonitoring programs to manage and conserve these threatened small lakes and reservoirs.

2.4. Fish and fisheries in lakes: composition, abundance, catch and effort

Fish are essential natural resources that contribute immensely to the economy of many countries worldwide. Fish provide an affordable source of animal protein to thousands of millions of less privileged people worldwide (Stankus, 2021). Fishes are the most diverse and widespread among aquatic organisms, occurring in almost all water masses (Lévesque *et al*., 2007). The online fish species repository in Fish Base (Froese & Pauly, 2018) has at least 33,000 species that have been described, representing more than the combined total of all other vertebrate species on earth. Knowing the species diversity, abundance and distribution of fishes in different waterbodies can provide insight into the integrity and status of the ecosystem (Thompson *et al*., 2015), which helps to develop sound management and conservation programs.

Studies in tropical regions suggest that species richness and fish community composition are spatially and temporally structured (Saint-Paul *et al*., 2000). Spatial and temporal patterns of diversity, distribution and species composition of freshwater fishes help examine factors influencing the structure of the fish community (Galactos *et al*., 2004). Rapid environmental changes and human activities have altered biological diversity and associated ecosystem services or functional diversity across the globe (Mao *et al*., 2021). Among freshwater lakes and reservoirs, the major environmental factors that influence the distributions of species include water transparency, dissolved oxygen concentration, nutrient concentrations/ productivity and morphometry (depth, size, shape) (Rodrı´guez & Lewis 1997; Tejerina-Garro *et al*., 1998; Su'arez *et al*., 2001).

Aquatic trophic structures are driven by ecosystem size, productivity, and disturbance, ultimately influencing food web complexity (Jia *et al*., 2021). For instance, ecosystem size driver theory says that large ecosystems are associated with long food chains (more opportunities for trophic interaction) resultant from high habitat availability and diversity (Post *et al*., 2000), while productivity driver relates to species richness limits due to the rate at which energy enters a system ultimately affecting the food web (Wright, 1983). On the other hand, the ecosystem disturbance driver entails shorter food chains in highly variable environments (Pimm & Lawton, 1977). Therefore, the number of fish species increases with lake size (Jia *et al*., 2021), while an increase in chlorophyll-*a* concentration (an indirect measure of phytoplankton productivity) favours the occurrence of fish species (Currie *et al*., 2004). Anthropogenic activities that disturb the ecosystem, such as nutrient loading, habitat loss and fragmentation, resource overexploitation, and climate change, affect biodiversity (Dudgeon *et al*., 2006; Hastings & Wysham, 2010; Wang *et al*., 2015; Amoutchi *et al*., 2021; Yongo *et al*., 2021).

Fish response to environmental drivers is species-specific, leading to spatio-temporal variations in fish assemblages (Kolosa, 1989). For example, in open water, zooplankton, insect larvae, and crustaceans attract specific fish species (Chan *et al*., 2020). Furthermore, fish abundance and biomass are also influenced by fluctuations in water levels; increasing water levels are associated with more catches due to increased littoral zone used for feeding and the overall increase in habitat (Walumona *et al*., 2022; Morara *et al*., 2022). Fish stocking, a common fisheries management tool, also strongly influences lake fish communities, i.e., filter-feeding and benthivorous fish stocking seldomly suppress phytoplankton, instead promote them while predators alter nutrient cycles resulting in fish community changes (Mao *et al*., 2021). In addition, fish traps at the river mouths during fish migration result in catching mature fish before spawning, which affects recruitment, and lead to low catches (Outa *et al*., 2020). Persistent and increasing environmental changes and human activities threaten freshwater biodiversity; hence understanding the linkages between biological communities and their environment is essential.

Fishes are prominent biological indicators of ecosystem integrity since many species respond differently to water and habitat quality changes and other human influences (Karr *et al*., 1986; Zainudin, 2005; Hamzah, 2007; Achieng *et al*., 2021). López-López & Sedeño-Díaz (2015) notes that fish, as long-lived water dwellers, continually integrate the chemical, physical and biological histories of the aquatic ecosystems, making them good bioindicator of current and long-term water quality. Besides, fish are sensitive to several kinds of disturbance and stressors, such as hydrologic alteration and the impact of pollutants (López-López & Sedeño-Díaz, 2015). Fish communities can also be used to evaluate societal costs of degradation because their economic and aesthetic values are widely recognised (Fausch *et al*., 1990). Considering that freshwater ecosystems are the most threatened habitat types in the world (Dudgeon *et al*., 2006), the occurrence, distribution and population dynamics of fishes and other aquatic organisms have become important and popular subjects of study (Tolonen *et al*., 2005; Haddad *et al*., 2015; Bryan-Brown *et al*., 2020).

Human population growth has exerted enormous pressure on natural resources resulting in biodiversity loss or declines in many ecosystems worldwide (Moyle & Leidy, 1992; DeFries *et al*., 2004 Darwall *et al*., 2018; Tickner *et al*., 2020). Many small waterbodies in the tropics are fringed by extensive wetlands that are often targeted for development and use for tourism and agricultural activities because of their rich biodiversity and fertile soils, respectively (Aloo, 2003; Dahlberg & Burlando, 2009; Yu *et al*., 2018). In the Lake Victora basin, many wetlands fringing Lake Victoria, satellite lakes and river mouths of the major rivers have been lost, degraded or converted to other uses such as grazing, farming, agroforestry and human settlement (Kairu, 2001; Aloo, 2003; Thenya *et al*., 2006; Masese *et al*., 2012; Okeyo-Owuor *et al*., 2012; Rongoei, 2003).

It is estimated that wetlands occupy <1% of the earth's surface, but they are among ecosystems that support high biodiversity and provide habitat for about 20% of the world's species (Dugan, 1993). Fish diversity in wetlands is also affected by accidental and intentional exotic introductions that outcompete native fish species. Alien fish species have been widely introduced globally for biological control, as bait for certain capture fisheries and aquaria, and to boost fish production for food (Canonico *et al*., 2005). Although there are many nuisance alien species, notably African catfish (*Clarias gariepinus*), tilapia (*Oreochromis niloticus* and *O. mossambicus*), pangasius (*Pangasionodon hypophthalmus*), big head (*Hypophthalmichthys nobilis*), pacu (*Piaractus brachypomus*), common carp (*Cyprinus carpio*), suckermouth catfish (*Pterygoplichthys* spp.), rainbow trout (*Oncorhynchus mykiss*) etc. are widely distributed in Asia (Singh & Lakra, 2011; Xiong *et* *al*., 2015; Hossain *et al*., 2016; Joshi *et al*., 2021), North and South America (Gido & Brown, 1999; Santos *et al*., 2018), and Europe (Zenetos *et al*., 2009; Grabowska *et al*., 2010; Emiroğlu, 2011; Piria *et al*., 2018) threatening the stability of freshwater ecosystems and the survival of the native ichthyofauna and overall aquatic biodiversity (Piria *et al*., 2018). Likewise, invasive fish species such as tilapia (*Oreochromis niloticus, O. mossambicus, O. aureus*, and *Coptodon zillii*), common carp (*Cyprinus carpio*), Rainbow trout (*Oncorhynchus mykiss*) and Brown trout (*Salmo trutta*) have widely been introduced across Africa (Sultana & Hashim, 2015; Weyl *et al*., 2020).

Kenya also has invasive alien fish species introduced into Lake Baringo, Naivasha and Victoria, including six species (*Oncorhynchus mykiss*, *Salmo trutta*, *Cyprinus carpio*, *Gambusia affinis* Girard, *Lates niloticus* and *Micropterus salmoides*) featured in the list of "100 of the World's Worst Invasive Alien Species" (Hickley *et al*., 2008; Outa *et al*., 2020). In the Lake Victoria basin, the introduction of Nile Perch (*Lates niloticus*) in Lake Victoria (Barel *et al*., 1985; Kudhongania *et al*., 1992; Goudswaard *et al*., 2002) poses a threat to satellite lakes, such as Lakes Sare and Kanyaboli due to the interconnectedness between the systems (Gichuki *et al*., 2005; Abila & Othina, 2005). Likewise, *Oreochromis niloticus* has established itself in Lake Kanyaboli and most of the Yala Wetland system, threatening the existence of native tilapiine species due to potential hybridisation with *O. esculentus* and *O. leucostictus* (Nyingi & Agnèse 2007; Angienda *et al*., 2011; Ndiwa *et al*., 2014) and competition (Katunzi, 2004; Mwanja *et al*., 2012).

The Yala Wetland is one of the important wetlands in western Kenya because of the threatened Sitatunga antelope (*Tragelaphus spekeii*), several bird species, and the threatened *Oreochromis esculentus* and *O. variabilis* (Aloo, 2003; Abila 2005). A significant portion of the wetland has undergone reclamation for agriculture. This has negatively influenced the ecosystem integrity of the wetland and the satellite lakes (Lakes Kanyaboli, Sare and Namboyo) therein (Wilfred *et al*., 2005; Abila & Othina, 2005). Lake Kanyaboli is the biggest among the three lakes and has been the most affected by the reclamation of the Yala Wetland and other human activities in its catchment (Aloo, 2003; Abila, 2005; Angienda *et al*., 2011; Kondowe *et al*., 2022a). Nevertheless, the lake has a rich diversity of fish species, including *Oreochromis esculentus* and *O. variabilis* that have disappeared in Lake Victoria, and six haplochromine species, some of which have also disappeared in Lake Victoria and are classified as threatened by the IUCN Red List (IUCN, 2019) among others, e.g., *Coptodon zillii*, *Clarias gariepinus*, *Protopterus aethiopicus*, and *Xenoclarias* spp. (Aloo, 2003; Gichuki *et al*., 2005).

Although Lake Kanyaboli is a refuge for remnant species of Lake Victoria, the lake's fish and fishery are threatened by overexploitation by the increasing number of fishers and organic pollution from the catchment. Biomonitoring studies on the lake are limited and infrequent, and most of the data available predate the major developments and reclamation efforts in the Yala Wetland and around the lake (Okemwa, 1981; Mavuti, 1989; Aloo, 2003; Kondowe, 2022a). Moreover, the previous studies concentrated on the occurrence of various species (Maithya, 1998; Aloo, 2003; Masai *et al*., 2005) with a limited estimation of fish catch and effort. This study sought to fill these gaps by describing the historical and present status of fish species diversity, composition and abundance and the drivers of their temporal variation in the lake. I hypothesised that changes in fish composition, diversity and abundance in Lake Kanyaboli and similar small waterbodies in the tropics result from fishing pressure and anthropogenic activities in the catchment areas.

The commonly used index in fisheries management is the catch per unit effort (CPUE), with a proportional change in CPUE representing the same proportional change in stock size (FAO, 1997). The CPUE is a function of fish abundance and catchability of the fishing gear (Linlokken & Haugen, 2006), and it is often used as a measure of fish abundance (Olin *et al*., 2004) and species composition (Olin *et al*., 2004). The CPUE expresses the quantity of fish caught by a given amount of fishing effort. When an increase in the fishing effort leads to a decrease in the catch, the population may be overfished (Shirakihara *et al*., 1992).

The fishing effort is categorized into either nominal or effective fishing effort (Zeller *et al*., 2021). The nominal effort reflects the simple total of effort units exerted on a stock in a given period, while effective effort is corrected based on differences in fishing power and efficiency and relates to a specific fishery and gear (FAO, 2019). According to FAO (2019), if more than one gear is considered, standardization in relation to one of them is necessary. The determination of the best measure efforts depends on the objectives being pursued, for example, a good measure of fishing effort for biologists should be proportional to fishing mortality, while for economists, it should be proportional to the cost of fishing (FAO, 1997).

Thus, the method of catch and effort data collection and analysis should be specified in line with the measure of fishing effort chosen.

Furthermore, fish abundance is also a function of recruitment which is influenced by environmental conditions such as wind and wave action, particularly strong wind and waves result in high egg and larval mortality (Clady 1976; Aalto & Newsome, 1993; MacKenzie 2000; Weber & Brown, 2013). Increased turbulence and turbidity in shallow lakes with fine sediments due to windy conditions also alter the visual environment, making foraging difficult (Weber & Brown, 2013). Plant species composition, distribution, and percentage cover influence the fish species composition, individual fish species production, access to fish stocks by fishers, fishing gear, and sometimes boat access and transport possibilities for getting the fishery product to the markets (Peter, 200). Macrophytes influence the zooplankton community by contributing to increased species richness (Dos Santos *et al.*, 2020) besides providing substrate for colonisation, refuge, feeding, and spawning grounds for fish which boosts production (Gasith & Hoyer, 1998). Despite the ecological importance of macrophytes, moving islands (made of floating macrophytes) in Lake Kanyaboli interfere with fishing by blocking landing sites and sweeping away fishing gears (gillnets and longlines) hence considered a nuisance by fishers.

Historically, small-scale artisanal and subsistence fisheries have not received attention until recently because they were considered temporary features during an anticipated transition to industrial fisheries (Zeller *et al.*, 2021). Nevertheless, the sector continues to experience challenges due to a lack of institutional and political support, which is skewed toward large-scale (i.e., industrial) fisheries because of the perceived higher direct macroeconomic contributions to national or government income (Zeller & Pauly, 2019). Therefore, monitoring studies on fish and fisheries in Lake Kanyaboli are limited. The available ones were done before the maintenance of the influent canal, retention dyke, and outlet; hence do not capture the after-effects (Okemwa, 1981; Mavuti, 1989; Aloo, 2003). Moreover, previous studies concentrated on the occurrence of various species (Maithya, 1998; Aloo, 2003; Masai *et al*., 2005) but not on temporal changes in abundance, catch, composition and CPUE besides being short-term studies. Therefore, the long-term generation of fish species composition, abundance and catch data is vital. These data are also crucial for informing biomonitoring and conservation efforts of the threatened fish species and fishery that many people depend upon.

Nevertheless, the availability of catch and effort data in Lake Kanyaboli is a challenge because of the lack of detailed studies and the small-scale nature of the fishery, which is associated with open access and under-reporting of catch and effort by fishers. Likewise, these challenges have been reported in other small-scale fisheries elsewhere due to the under-reporting and migratory nature of the fishers (Zeller *et al*., 2021; Peel *et al*., 2015). This results in a lack of detailed data to analyse and estimate parameters used in fisheries management, such as catch per unit effort (CPUE). However, standardized measure of effort, such as effective fishing effort based on the power input in fisheries, i.e., kWday (Greer *et al*., 2019), is timely for data-poor fisheries such as Lake Kanyaboli fishery.

2.5. Human impact on physico-chemical variables and ecology of lakes

Many inland water bodies have witnessed considerable degradation due to the loading of nutrients and organic matter from catchment areas leading to eutrophication and changes in plankton communities (Wang *et al*., 2015; Abdul *et al*., 2016). Pollutants entering the lake include semi-treated municipal sewage, animal faecal waste, and agrochemical surface run-off from farms in the catchment (Otiang'a-Owiti & Oswe, 2007). Many tropical lakes are now considered eutrophic, with blue-green algae blooms and microcystins threatening water quality and fish production (Sekadende *et al*., 2005; Ndebele & Magadza, 2006; Paerl *et al*., 2011; Onyango *et al*., 2020; Díaz-Torres *et al*., 2021). Fish kills and the temporary shutdown of the drinking water supply have been reported in Lake Victoria due to cyanobacterial blooms in the Nyanza Gulf (Simiyu *et al*., 2018). The lake also shows shift in its phytoplankton community due to P enrichment and depletion of S_i i.e. from diatom and chlorophyte dominance to dominance by filamentous and colonial cyanobacteria (Hecky *et al*., 2010). The microcystins affect fish from the embryonic stage through to adult life by influencing hatching, survival, growth rates, fish activity, neurotoxicity risk, osmoregulation, liver activities, and heart rates, as well as histopathological (Rodger *et al*., 1994; Malbrouck & Kestemont 2006; Mitsoura *et al*., 2013; Yu *et al*., 2021).

With the increasing influence of human activities on inland waters, there is a growing need to understand the linkages between water quality and biological communities to implement management measures. Water quality challenges are likely to increase in reservoirs and lakes receiving high influents (nutrients and particulate matter) associated with anthropogenic activities (Mustapha, 2009; Ling *et al*., 2017; Lee & Oh, 2018). The interactions between primary producers and secondary consumers can also influence water quality (Friedland *et al*., 2012). For instance, reducing the stocks of planktivorous fishes enhances the survival of zooplankton (Jeppesen *et al*., 2007; Gulati *et al*., 2008; Barnes *et al*., 2015). This reduces planktonic algae abundance, which serves as food for the zooplankton (Barnes *et al*., 2015). The result is improved water quality due to reduced phytoplankton biomass contributing to turbidity and subsequent poor trophic status. Similarly, water quality changes and biological communities' fluctuations are likely in tropical lakes due to agroindustry and biomass extractive practices in catchments (Liu *et al*., 2011; Moncayo-Estrada *et al*., 2012; Kondowe *et al*., 2022a; Walumona *et al*., 2021).

For lakes and reservoirs, the characteristics of the drainage or catchment area (e.g., size, topography, land use, geology, etc.) and the size of the recipient ecosystem (e.g., river, lake, or reservoir) control their trophic status (Walker *et al*., 2007). Therefore, activities that increase the amount of nutrients in the surface runoff into the lake speed up the process of eutrophication. In addition, land use and land cover changes related to population increase are also among the drivers of water quality and lake biological community changes in inland waters (Catherine *et al*., 2013; Kibena *et al*., 2014; Bucak *et al*., 2018; Leech *et al*., 2018; Namugize *et al*., 2018).

Inland waters (particularly shallow lakes and reservoirs) in tropical and subtropical regions are associated with changing plankton assemblages due to seasonal wind speed and rainfall changes, which affect water mixing and influent nutrient (Geraldes & Boavida, 2004; Zhu *et al*., 2014). These (wind speed and rainfall) also influence Secchi depth (James & Chimney, 2008) and water residence times (Geraldes & Boavida, 2004; Borges *et al*., 2008; Srifa *et al*., 2016; Ismail *et al*., 2019). Furthermore, Cardille *et al*. (2004) noted that lakes with high ratios of the catchment to lake surface areas had shorter water residence times since such lakes cycle more water through the lake.

Water abstraction for irrigation, hydropower, domestic use, and other related anthropogenic activities also influence the water quality and aquatic life in lakes. The water demand to satisfy these developments and services (irrigation, hydropower and domestic) is rising due to the increase in the human population (Becht & Harper 2002). Other equally important services in the communities surrounding water sources that require water include vegetable farms, wildlife and livestock needs, and industrial demands such as geothermal power generation (Goldson, 1993). Various modelling methods have been developed and tested to calculate water requirements for different local and industrial needs (Muvundja *et al*., 2009; Ding *et al*., 2014; Fowe et al., 2015). These models use the basic hydrological process of rainfall, river inflow, groundwater flows, evaporations, and river outflows to estimate mass balances in the lake to determine the ecological effects of water abstraction (Cardille *et al*., 2004; Beaton *et al*., 2007; Muvundja *et al*., 2009).

Lake Naivasha estimates show surface water abstraction is between 46.4 and 60 million cubic metres per year (Mmbui, 1999, Becht & Harper, 2002). Shallow lakes and reservoirs are more susceptible to evaporation, but recently water abstraction for various uses has exacerbated the problem; hence, inflows and stable rainfall regimes are critical to maintaining good water quality (Jarosiewicz & Witek, 2014; Fowe *et al*., 2015).

Moreover, lakes such as Lake Kanyaboli are targets for tourism and coastal development, which disturbs the aquatic environment if not properly planned and managed. It is vital to holistically look at the human impacts on the well-being of lakes to ensure proper management of these ecosystems for continued ecological and economic benefits.

CHAPTER THREE

MATERIAL AND METHODS

3.1. Description of the study area

The present study was carried out in Lake Kanyaboli on the northern shores of Lake Victoria (Figure 2). The lake has an area of 10.5 km^2 and a 3 m average depth (Abila, 2005). Thus, the lake has an estimated volume of 3.15 × 107 m^3 (0.0315 km^3). Lake Kanyaboli is part of the extensive papyrus-dominated Yala Wetland, which fringes the northern shores of Lake Victoria. The wetland also contains two other lakes, Lake Sare and Lake Namboyo. Lake Sare has an area of 5 km^2 and an average depth of 5m, while Lake Namboyo covers an area of 0.01 km^{2,} and the depth averages 17m (Angienda *et al.*, 2011). The largest of the three lakes in Yala Wetland, Lake Kanyaboli, is located between lies between latitudes 0◦05S'N and 0◦02'N and longitudes 34◦09'E and 34◦ll'E (Abila *et al.*, 2004, 2008; Opiyo and Dadzie, 1994).

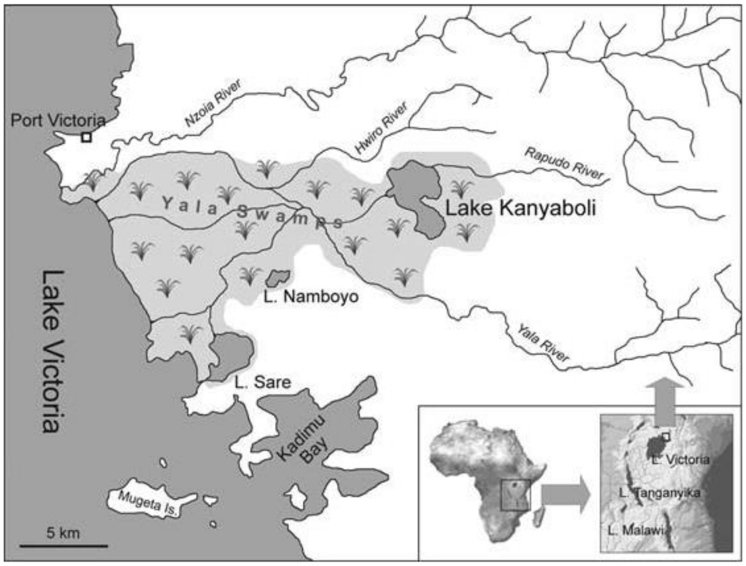

Figure 2: Map of the Yala Wetland with Lakes Kanyaboli, Namboyo, and Sare (source: Abila *et al*., 2008).

Before the 1970s, Lake Kanyaboli used to be replenished by precipitation, water from Lake Victoria back-flow and Yala Wetland through water drained from the Yala River (Wilfred *et al*., 2005). The lake was directly connected to the wetland before a dyke was constructed across the lake to reclaim 2,300 ha for agriculture (Owiyo *et al*., 2014). After Lake Kanyaboli was separated from its surrounding wetland, a 9-kilometer-long influent canal was built to reconnect it to the Yala River. However, between the years 1980 and 2005, the influent canal was non-operational because it had become clogged by communities that used it to water their livestock (Mavuti, 1989; Aloo, 2003; Abila, 2005). The canal, together with the dyke and outlet, was maintained by Dominion Group of Companies, which was given a lease in 2003 for agroindustry in the reclaimed part of Yala Wetland (Burgess, 2008). The influent canal has an estimated discharge rate of 5 m^3s^{-1} at maximum capacity (Burgess, 2008). Thus, the current sources of water for the lake include the Yala River influent canal, rainwater, and backwash waters from Lake Victoria through the Goye causeway (Wilfred *et al*., 2005). Lake Kanyaboli experiences water quality changes related to activities in the Yala Wetland catchment and riparian areas (Kondowe *et al*., 2022a). The major human activities in the lake are fisheries and harvesting of papyrus for handcraft and fuelwood. On the other hand, the reclaimed Yala Wetland is used for subsistence and mechanised agriculture, pasture for livestock, as well as harvesting of papyrus and wood for handcraft and fuelwood.

The reclaimed part of the Yala Wetland was converted into a commercial agricultural site through initiatives by the Government of Kenya. This culminated in large-scale farm activities by a private company called Dominion Group of Companies in 2004 (Kinaro, 2008). Dominion Farm commissioned an environmental impact assessment (EIA) for largescale rice production, and the company was issued a license in 2004 (Owiyo *et al*., 2014). Instead of growing rice on the reclaimed 2,300 hectares, as was originally planned, the company diversified into other types of agriculture and development activities (Abila, 2005). The activities included the construction of irrigation dykes, canals, dams and weirs, an airstrip, and a road.

Furthermore, the project was also involved in an aquaculture venture, which included fish farms, fish processing and fish meal factories (Abila, 2005). The expansion of activities by Dominion Company led to the reclamation of more wetland areas initially occupied by local communities (Owiyo *et al*., 2014). This was a catalyst for a cascade of events that resulted in a disagreement between the company and stakeholders (including the community) on wetland conservation and the needs/ interests of the community. The continued disputes between the company and stakeholders led to Dominion Farms Ltd.'s winding up in 2017 despite having a 25years lease since 2003.

The Yala Wetland region has bi-modal rainfall, with a long rainy season from March to June and October to December is the short rainy season. The climatic conditions of the area are influenced by the movement of the inter-tropical zone and the proximity of Lake Victoria. As a part of the lake-shore belt, Lake Kanyaboli receives an average annual precipitation of less than 1300 mm (Anyona, 1997) (Table 1). Constant winds also blow on the lake, which averaged 3.14 ms^{-1} between February 2021 and April 2021 as observed from Deeper Smart Sonar Pro+ (ITGAM0303, Deeper, UAB, Antakalnio, Lithuania). Other characteristics of the lake have been summarized in Table 1.

Table 1: General features of Lake Kanyaboli, Kanya

Attribute	Value	Source
Volume (Area * depth) (km^3)	0.0135	Present study
Mean depth (m)	3	Abila, 2005
Lake area (km^2)	10.5	Abila, 2005
Catchment area (km^2)	175	Odero & Odenyo, 2021
Precipitation over lake surface (mm)	1,300	Anyona, 1997
Lake surface evaporation rate (mm)	1800	Akwiri *et al*., 2016

Yala Wetland's three lakes (Kanyaboli, Sare, and Namboyo) are ecologically significant because they are home to Lake Victoria's diverse cichlid fish fauna (Abila *et al*., 2004, 2008; Gichuki *et al*., 2005). The cichlids include haplochromines (*Lipochromis maxilaris*, *Astatotilapia nubilus*, *Astatotilapia* sp. *"big eye"*, *Pseudocranilabrus multicolour victoriae*, *Xystichromis phytophagus* and *Astatoreochromis alluaiudi*), *Oreochromis esculentus* and *O. variabilis* some of which have virtually gone extinct in Lake Victoria (Abila, 2005). The exotic Nile Tilapia (*O. niloticus*) has also established itself in Lake Kanyaboli. Other fish species reported in the lake include *O. leucostictus*, *Coptodon zillii*, *Clarias gariepinus*, *Protopterus aethiopicus*, and *Xenoclarias* spp (Aloo, 2003).

Lake Kanyaboli, together with Lake Sare, supports semi-commercial fishing. In contrast, fishing hardly takes place in Lake Namboyo because of the thick mats of macrophytes (*Cyperus papyrus* and *Phragmites australis*) that make the lake inaccessible (Wilfred *et al*., 2005; Abila & Othina, 2005). The phytoplankton community in Lake Kanyaboli is represented by over 40 species, and it is dominated by blue-green algae (Wilfred *et al*., 2005). Lake Kanyaboli fish catches continue to decline due to overfishing and poor fishery management, although the lake is an important source of livelihood for riparian communities through fisheries (BN Kondowe, unpublished data).

Yala Wetland plays an important role in filtering and buffering the waters draining from Yala River into Lake Kanyaboli and Victoria (CGS, 2015). For example, Muoria *et al*. (2015) reported that nitrate levels of water entering Lake Victoria were 3.61 mg L^{-1}, despite parts of the wetland recording as high as 9.84 mg L^{-1} attributed to buffering effect of the wetland. Therefore, human activities in the Lake Victoria catchment affect not only the lakes in the

Yala Wetland but also Lake Victoria, and reciprocally Lake Victoria influences the three satellite lakes (Kanyaboli, Sare, and Namboyo) through back-flow (Wilfred *et al*., 2005).

3.2. Data sources

Primary and secondary data were used in the present study. The primary data collected included physical, chemical, and biological variables such as pH, electrical conductivity, dissolved oxygen, temperature, Secchi depth, nitrates, nitrite, ammonium, soluble reactive phosphorus, total nitrogen, and total phosphorus and chlorophyll-*a*. The data were collected monthly from February 2020 to February 2021, except chlorophyll-*a* data that were collected for six months (June 2020 to November 2020). The twelve months sampling period covered the dry season (January 2021, February 2020 and 2021, and July to September of 2020) and wet season (March to June 2020 and October to December 2020). Similarly, primary data on fish abundance, composition, and catch, as well as fishing effort, were also collected monthly from February 2020 to February 2021.

Primary data was supplemented by secondary data on physical and chemical variables, chlorophyll-*a*, fish abundance, composition, and catch and fishing effort from published and grey (theses and reports) literature. However, studies on the limnology and trophic status of Lake Kanyaboli are limited or lacking. Thus, studies from other Kenyan lakes (Lake Baringo, Lake Naivasha, and Lake Victoria), African lakes (Lake Tana, Lake Bunyonyi, and Lake Nasser) and elsewhere (Lake Akkulam–Veli, Perry Pond, meres of the North-West Midlands in the United Kingdom, and Lake Okeechobee) were also used to inform discussion and recommend management plans.

3.3. Sampling sites

Six sampling sites were selected for monitoring surface water quality in Lake Kanyaboli, representing fish landing sites and villages around the lake (Figure 3). These were in the littoral zone (15 m from the shore) (stations 1, 3, 4, 5, and 6) and limnetic zone (open water) (station 2) of Lake Kanyaboli. The six sampling sites from the lake were also used for sampling phytoplankton and zooplankton for assessing plankton communities. The six were Station 1 = Gangu, Station 2 = Open Water, Station 3 = Prince’s Hotel, Station 4 = Kadenge, Station 5 = Inlet from the lake-side and Station 6 = Kombo Beach (Outlet from the lake-side). Besides these six stations, water quality variables and discharge rate were sampled from the inlet canal before the water entered the lake and outlet after the water left the lake (Figure 3). The inlet canal was sampled for nutrients to determine the nutrient concentration used in the mass balance equation for the estimation of mass loading. On the other hand, the outlet was sampled to estimate the nutrient loads leaving the lake.

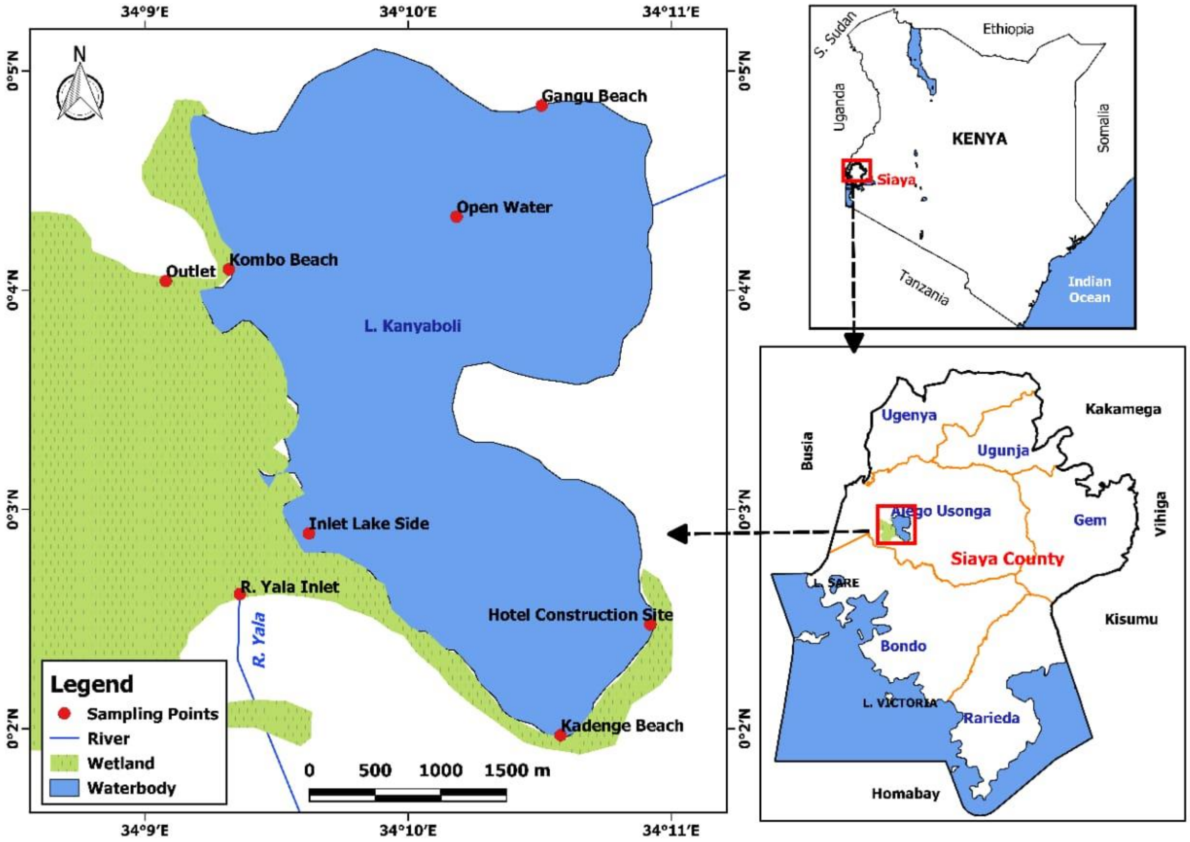

Figure 3: Map showing six sampling sites in Lake Kanyaboli, inlet and outlet from February 2020 to February 2021

3.4. Sample collection and analysis

3.4.1. Objective 1: Historical changes in the limnological characteristics and trophic status of Lake Kanyaboli from 1981 to 2020

Water quality variables, including temperature, pH, Secchi depth, dissolved oxygen, and electrical conductivity, were measured *in situ* at all the sampling sites using a YSI multi-probe water quality meter (556 MPS, Yellow Springs Instruments, Ohio, USA). All variables were measured in triplicates every month from February 2020 to February 2021. The filterable nutrients, namely ammonium (NH_4^+), nitrate (NO_3^-), nitrite (NO_2^-), and soluble reactive phosphorus (SRP), were determined in the laboratory at the University of Eldoret. These were determined from surface water samples collected from all the sampling sites. The water samples used for filterable nutrients determination were filtered immediately in the field using pre-combusted GF5 Whatman filter papers (47 mm diameter; 0.7 µm pore size) into triplicates of sampling bottles placed in an icebox for transportation to the laboratory. Filter papers were also kept and used for analyzing chlorophyll-*a*. The filter papers were placed in an icebox in a dark container for transportation to the laboratory. Furthermore, unfiltered water samples were collected in triplicate and used for analyzing total phosphorus (TP) and total nitrogen (TN). Preservation and transportation of unfiltered water samples for TP and TN analysis were done as those for filterable nutrients.

The analysis of NH_4^+, NO_3^-, NO_2^-, SRP, TP and TN in the laboratory was done following standard colourimetric methods (APHA, 2017). The nutrient concentrations were calculated from absorbances using equations in Appendix I (a – c and e). The water quality variables, including SRP, NH_4^+, NO_3^-, and NO_2^-, were analyzed using the molybdenum blue method, dichloroisocyanurate – salicylate method, cadmium reduction method, and azo – dye complex formation, respectively. On the other hand, the ascorbic acid reduction method and the persulfate digestion method were respectively used for TP and TN. The nutrient concentrations were calculated from absorbances using equations in appendix I (d and f). In addition, pigment extraction using 90% acetone was used to analyse chlorophyll-*a* concentration following APHA (2017).

Chlorophyll-*a* was extracted using 95% acetone and centrifugation for 15min at 4,800 rpm in a centrifuge before analysis (Swanepoel *et al*., 2008). The chlorophyll-*a* absorbance was measured using a spectrophotometer at 750 and 665 nm. At the same time, acetone (90%) was used as a blank. Equation 1 (Yang *et al*., 2007) was used to calculate the concentration of chlorophyll-*a*.

$$\text{Chl-}a\ (\mu g/L) = [27.3 \star (A_b665 - A_b750) - (A_a665 - A_a750)] \star V_1/V_2 \qquad (1)$$

Where:

Subscripts = absorbance before acid addition (Ab) and after acidification (Aa).

V_1 = volume of solvent (ml).

V_2 = volume of sample (ml).

3.4.2. Objective 2: Factors that influence nutrient (nitrogen and phosphorus) dynamics in Lake Kanyaboli using a mass balance model

3.4.2.1. Sample collection

The input and output sources of water and nutrient budgets of Lake Kanyaboli (Figure 4) were sampled for analysis of water and nutrient budgets. The inputs for water balance include surface flow (influent canal), precipitation and ground inflow, while output includes outflow in the outlet, groundwater outflow, evaporation and evapotranspiration. Likewise, nutrient sources include surface influent, groundwater inflow, atmospheric deposition (dry and wet deposition), remineralization from sediments and nitrification (for nitrogen). In contrast, nutrients in a lake are lost through surface outflow, groundwater outflow, sedimentation and denitrification (for nitrogen).

Although groundwater inflow and outflow could not be estimated in the current study, an assumption was made that input from groundwater was important because the lake survived a prolonged lack of inflow (between the 1980s and 2004) without either showing major shrinkage or completely drying up (Okemwa, 1981; Anyona, 1997; Aloo, 2003; Abila, 2005). The major sources of water when the influent canal was blocked were rainfall and an assumed seepage (Mavuti, 1989; Abila, 2005). Similarly, the present study could not estimate nutrient input from groundwater inflow, remineralization and nitrification. Nutrient losses in groundwater and denitrification were also not estimated. However, assumptions and comparisons from other lakes and reservoirs were used to infer the importance of these sources and losses for Lake Kanyaboli.

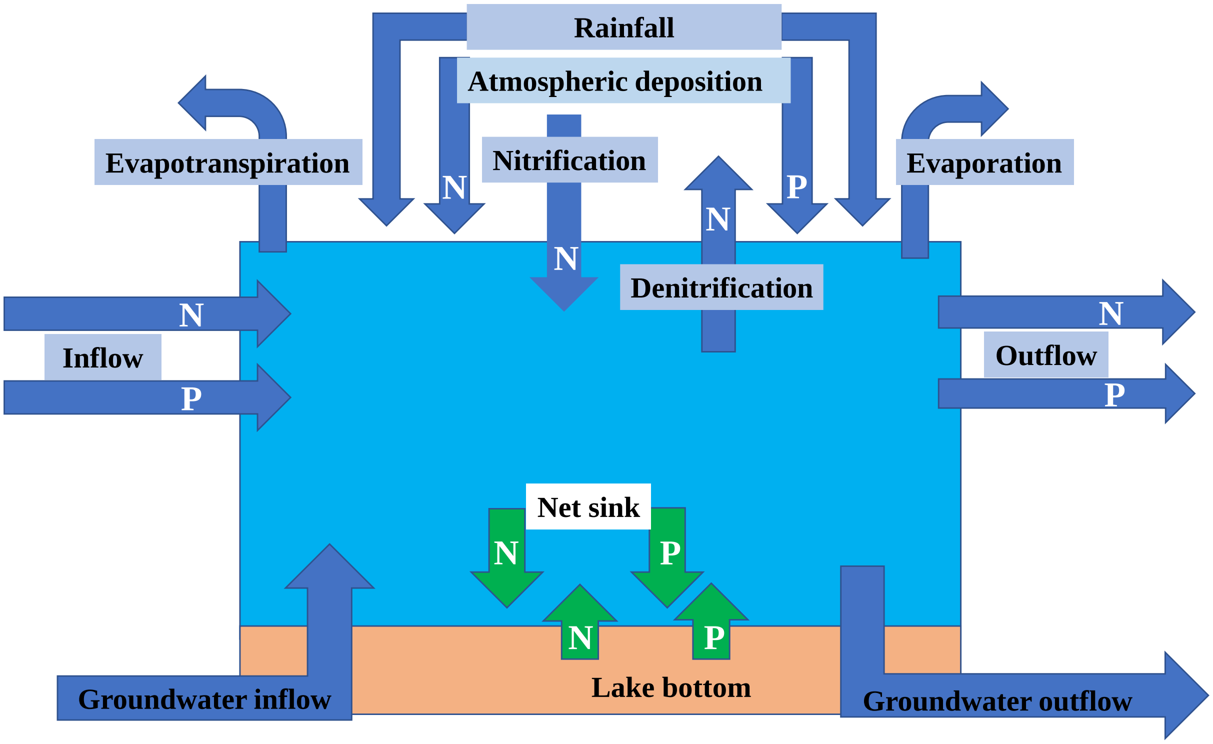

Figure 4: Conceptual diagram showing the water and nutrient sources and losses in Lake Kanyaboli

3.4.2.1.1. Water discharge sampling from Influent canal and outlet

The influent canal and outlet were sampled for discharge after the sluice gates from February 2020 to February 2021, except for May 2020, to estimate water input and output budgets in Lake Kanyaboli. Discharge was estimated by multiplying the velocity of the water by area (width multiplied by depth). The velocity-area method was used to calculate velocity (Wetzel & Likens, 2013). Five transects were used to determine the inlet canal and outlet width, while water depth at each transect was measured using a 1 m ruler.

3.4.2.1.2. Rainfall and evaporation sampling

Rainfall and evaporation are also important components of lake water balance. The present study did not measure these two parameters instead, secondary data was used. The annual rainfall of lake Kanyaboli is estimated at 1,100 mm and 1500 mm (Anyona, 1997; Okungu *et al*., 2005; Mwakubo & Ikiara, 2008), while the evaporation rate is estimated between 1800 mm and 2000 mm (Akwiri *et al*., 2016). Besides rainfall and evaporation, evapotranspiration is another source of water loss for lakes fringed by dense macrophytes, such as Lake Kanyaboli. However, evapotranspiration accounting for water loss from macrophytes fringing the lake was not estimated. Nevertheless, the influence of evapotranspiration was considered when interpreting and discussing the results of water balance using results from papyrus-fringed Lake Naivasha in Kenya (Åse, (1987; Jones & Humphries, 2002) and Kirinya West Wetland in Uganda (Saunders *et al*., 2007).

3.4.2.1.3. Nutrient sampling from influent canal and outlet

Surface water samples were taken on a seasonal basis comprised of wet (March 2020 to June 2020 and October 2020 to December 2020) and dry seasons (January 2021, February 2020 and 2021, July, August, and September 2020) for nutrient concentration analysis. The nutrient analysed included ammonium (NH_4^+), nitrate (NO_3^-), total nitrogen (TN), soluble reactive phosphorus (SRP) and total phosphorus (TP). The water samples were collected, treated and analysed as described in section 3.4.1.

3.4.2.1.4. Atmospheric deposition

Atmospheric deposition samples were collected at the Kadenge sampling station following the method of Muvundja *et al*. (2009). Small plastic buckets cleaned with distilled water with an area of 4.87 m^2 were used for wet deposition by exposition to the open air at 2 m above ground at the beginning of a rain event. The same containers were used for dry deposition but filled with a pre-defined volume of 1 L of distilled water. The containers were exposed for ~24 h to open air at 2 m above ground. The initial and final sample volumes and the exposition time were recorded. Dry deposition sample collection was done during dry days, and any rain-contaminated samples were discarded. The water samples were preserved and analysed the same way as lake water samples in section 3.4.1. The atmospheric deposition analyzed six samples each for wet and dry deposition.

3.4.3. Objective 3: Spatio-temporal dynamics in environmental conditions and how they drive plankton communities in Lake Kanyaboli

3.4.3.1. Sample collection

Other than physico-chemical variables sampling (objectives 1), water samples were simultaneously collected for phytoplankton and zooplankton community analysis. Surface water samples were collected using a plankton net and transferred into sampling bottles. The samples were fixed using 10% Lugol's solution onboard before transportation to the laboratory for plankton (phytoplankton and zooplankton) analysis.

The sedimentation method following Swanepoel *et al*. (2008), Intergovernmental Oceanographic Commission of UNESCO (2010) and Soares *et al*. (2011) was used to process phytoplankton and zooplankton for identification and enumeration. A 10 mL sample was placed in a measuring cylinder and allowed to settle by gravity for 24 hr. Then excess water was siphoned off before phytoplankton and zooplankton taxonomic identification (species composition) and counting (abundance) (Swanepoel *et al*., 2008). Afterwards, the settled sample was homogenized, from which replicate samples of 1 mL were pipetted to slide (covered with coverslip) and examined under an Olympus inverted microscope (CK40, Japan) at ×40 magnification to identify plankton species and enumerate them. The sample on the slide was left for 20 min to settle before identification and enumeration. The abundances of phytoplankton and zooplankton were extrapolated to 1 L and expressed as individuals per litre (Ind/L). Planktons were identified to genus level using Prescott (1982), Bellinger & Sigee (2015) keys for phytoplankton, while Witty (2004) and Dang *et al*. (2015) keys were used for zooplankton. The identification was done to the genus level.

3.4.4. Objective 4: Temporal changes in fish species composition, catch and fishing effort in Lake Kanyaboli

3.4.4.1. Sample collection

Fish sample collection was done monthly from February 2020 to February 2021 at the beaches of Lake Kanyaboli. The collected samples were used for fish species composition, abundance catch and catch per unit effort (CPUE) data collection. Fishing effort data were also simultaneously collected. The lake was considered one unit for data analysis since it is not zoned into different fishing grounds due to its small size. However, all three landing sites, including Kadenge, Gangu and Kombo, were sampled as well as two other villages of Hawinga and Swila.

The fish catch, composition and effort survey targeted all fishers and the fishing boats they use with gillnet, traps and longlines. Fishers carry a combination of fishing gears when going out to fish, with gillnet being the most used either solely or in combination with other fishing gears (traps and longlines). The fishery is multispecies, with one fishing gear catching a combination of the following fish groups, tilapia, haplochromines, cyprinids, clarias and protopterus. The total catch by species for all the fishing boats operating during the study period was recorded by trained enumerators and then weighed in grams. The number of fish was also counted for individual species caught.

The monthly catches from the various fishing boats were pooled to estimate the total catch for each month. Fish samples collected were individually identified to species level where possible using keys and guides, such as Trewavas (1983), Witte & Densen (1995), Seegers *et al*. (2003) and FishBase (Froese & Pauly, 2018). Although there were two species of Clarias sp. (*Clarias gariepinus* and *Clarias liocephalus*), these were grouped and reported using the genus *Clarias* sp. Fish species composition data were collected for seven days each month for the whole study period. On the other hand, monthly fish weight (catch) was estimated using the daily mean catch calculated from seven days of fish weight data for each month for the entire study period.

The number of days fishers spent fishing was counted to estimate the number of days fishers were actively fishing between February 2020 and February 2021. Likewise, the active fishing boats operating in the lake were counted during the same period. In addition, February 2020 to February 2021 survey also used ten key informants comprised of two individuals from each designated landing site to collect data on the number of fishers, fishing crafts, type of fishing gear and fishing time. The data were used to estimate fishing efforts in Lake Kanyaboli. This helped track the temporal changes in fishing pressure and catches in the lake.

Data on fish composition, catches and effort were supplemented with secondary data from published and grey literature. However, the historical data were limited, infrequent and inconsistent, with only five years (2003, 2005, 2013, 2014 and 2015) having usable data for catch and effort, while four years (1981, 1998, 2003 and 2005) data were used for fish composition. But these data were only collected for short periods, <1 year. Furthermore, abundance data for different fish species were scarce; hence the secondary data was only used to estimate total (all species) catch and species occurrence.

3.4. Data treatment and analysis

3.5.1. Objective 1: Historical changes in the limnological characteristics and trophic status of Lake Kanyaboli from 1981 to 2020

Primary data on water quality variables (physical, chemical, and biological) were analyzed using descriptive statistics, Pearson correlation, and two-way ANOVA, which compared interaction between sampling sites and seasons. In cases where there was no interaction between sampling stations and seasons, one-way ANOVA was used, followed by Tukey's *post hoc* test for significant means among sampling stations for variables that showed spatial differences. Before analysis of variance (ANOVA), data were log-transformed to meet assumptions for parametric tests such as normality of the data. Furthermore, equality of variance was tested on original data using Levene's homogeneity test before pooling data from the six lake sampling sites.

The dimensionality of the physico-chemistry data was reduced using Principal Component Analysis (PCA) for both season and site. Two PCs were included to describe the spatial (site) and temporal (seasonal) changes in water quality variables. Permutational analysis of variance (PERMANOVA) was used to statistically assess PCAs based on Bray-Curtis similarity matrices (McArdle & Anderson, 2001).

3.5.1.1. Trophic status

Carlson trophic status index (TSI) equations based on Secchi depth transparency (SD), chlorophyll-*a* (Chl-*a*) and total phosphorus (TP) were used to calculate the trophic status of

Lake Kanyaboli (Carlson 1977). Furthermore, the overall trophic status of the lake was also computed using the Carlson Trophic State Index (CTSI) (Carlson and Simpson, 1996). The CTSI was an average TSI of the individual parameter values of SD, Chl-*a* and TP. The equations (2 to 5) used in the calculations were as follows:

$$\mathrm{TSI(SD)} = 10(6 - \ln \mathrm{SD}/\ln 2) \qquad (2)$$

$$\mathrm{TSI(Chl\text{-}}a) = 10(6 - 2.04 - 0.68 \ln \mathrm{Chl\text{-}}a/\ln 2) \qquad (3)$$

$$\mathrm{TSI(TP)} = 10(6 - \ln(48/\mathrm{TP})/\ln 2) \qquad (4)$$

$$\mathrm{CTSI} = \frac{[\mathrm{TSI\ (SD + TSI(Chl\text{-}}a\mathrm{) + TSI(TP)}}{3} \qquad (5)$$

Where:

SDD = Secchi disk depth in meters

TP = Concentration of Total phosphorus in milligrams per cubic meters

Chl-*a* = Concentration of chlorophyll-*a* in milligrams per cubic meters

TSI (SD) = Trophic state index based on Secchi disk transparency

TSI (TP) = Trophic state index based on total phosphorus

TSI (Chl-*a*) = Trophic state index based on chlorophyll

CTSI = Overall trophic index based on the average of equations 2, 3 and 4.

The TSI classifies lakes based on how green they are, which reflects the productivity of the lake (Brown and Simpson, 2001). The Carlson trophic index is a commonly used classification of trophic conditions of surface waters. Carlson trophic index (TSI) ranks lakes on a continuous numeric scale that ranges from 0 to 100 (Carlson 1977). The criteria for TSI indices classification for Secchi depth transparency (SD), total phosphorus (TP), and chlorophyll-*a* (Chl-*a*) are summarized in

Table 2.

Table 2: Criteria for classification of freshwater lake trophic status

TSI	**chlorophyll-*a* (µg/L)**	**Total phosphorus (µg/L)**	**Secchi depth (m)**	**Trophic status class**
<30 – 40	0 – 2.6	0 – 12	> 8 – 4	Oligotrophic
40 – 50	2.6 – 7.3	12 – 24	4 – 2	Mesotrophic
50 – 70	7.3 – 56	24 – 96	2 – 0.5	Eutrophic
70 – 100+	56 – 155+	96 – 384+	0.5 – <0.25	Hypereutrophic

(Source: Sheela *et al*., 2011)

3.5.2. Objective 2: Factors that influence nutrient (nitrogen and phosphorus) dynamics in Lake Kanyaboli using a mass balance model

The water balance of Lake Kanyaboli was estimated using equation 6 (Swenson & Wahr, 2009). The lake is data deficient in most parameters used for the water balance model, limiting long-term historical modelling; hence the current study only modelled one-year water balance based on data collected from February 2020 to February 2021.

$$dS/dt = P - E + Q_{in} - Q_{out} \quad (6)$$

Where:

dS/dt = the change in lake height with respect to time

P = the precipitation rate

E = the evaporation rate

Q_{in} = inflow to the lake from the surrounding catchment

Q_{out} = outflow from the lake.

The above water balance equation was re-written for Lake Kanyaboli based on the estimated parameters as follows:

$$dS/dt = P - E + Q_{influent\ canal} - Q_{outlet} \quad (7)$$

The volume of the lake was estimated as the product of the surface area of the lake and lake depth as measured in the current study. In addition, water residence time was also estimated as a quotient of lake capacity (volume) and inflow (influent and precipitation). The evaporation on the surface of the lake was accounted for by multiplying the lake surface area by the annual evaporation rate.

The nutrient balance was estimated following Ramírez-Zierold *et al*. (2010). Conceptually, Ramírez-Zierold *et al*. (2010) proposed that the mass budget for nutrients (phosphorus and nitrogen in this study) can be constructed by comparing summations of nutrient sources and losses as well as the net element flux due to overall internal processes (such as net sedimentation and net N_2 fixation-denitrification) using equation 8.

$$\Delta M_{(N, P)} / \Delta_t = \sum I_{(N, P)} - O_{(N, P)} \pm NIP_{(P, N)} \quad (8)$$

Where:

$\Delta M_{(N, P)} / \Delta_t$ = the change of N or P retained in the lake (or exported from the lake, if negative)

$I_{(N, P)}$ = the nutrient inputs from external (influent and atmospheric deposition) and internal (from sediment) sources

$O_{(N, P)}$ = the nutrient lost from the outlet.

$NIP_{(P, N)}$ = the net element flux due to overall internal processes derived from equation 8.

The above equation was re-written for Lake Kanyaboli and used for each nutrient (nitrogen and phosphorus) as shown below:

$$\Delta M_{(N, P)}/\Delta_t = \sum \mathbf{I}_{\text{influent canal}(N, P)} + \mathbf{I}_{\text{wet deposition}(N, P)} + \mathbf{I}_{\text{dry deposition}(N, P)} - O_{(N, P)} \pm NIP_{\text{net sedimentation}(P, N)} \quad (9)$$

Measurements of water balance components require various techniques, each associated with uncertainties (Fowe *et al*., 2015). The current study also considered uncertainty and errors resulting from equation 6 when estimating water balance components based on already established approximations. The errors in estimating the volume of rainfall and evaporation could also come from the uncertainties in the reservoir surface areas. The chosen errors for this study were associated with errors in measuring water input (10 %), including precipitation, inflow, and outflow, as well as atmospheric depositions (10 %) and discharge (Winter, 1981).

3.5.2.1. Nutrient load estimate

The nutrient load estimations were done following Muvundja *et al*. (2009). The transport of a nutrient (µg d^{-1}) is given by multiplying the riverine (inlet and outlet canal) concentration (µg m^{-3}) with the actual discharge (m^3 d^{-1}). The sampling of discharge was done once a month for twelve months (February 2020 to February 2021), which included the wet and dry seasons. The daily nutrient loads were then extrapolated to annual loads by multiplying the average concentration by the inlet and outlet canal's annual discharge (m^3 yr^{-1}).

Nutrient loads of dry deposition (µg m^{-2} d^{-1}) were obtained by:

$$C \cdot Vf \cdot (A \cdot \Delta_t)^{-1}$$

Where: C is the nutrient concentration (µg m^{-3}) in the bowl with the surface A (m^2) and the final water volume Vf (m^3). Δt is the time interval (d) of the exposition of the sample to the atmosphere. The average individual load estimate was multiplied by the lake area (10.5 km^2) and the number of dry days per year (201 days) to extrapolate to annual dry deposition over the entire lake (Opango *et al.*, 2005). The dry deposition on rainy days was assumed to be negligible (Muvundja *et al.*, 2009). The annual wet deposition loads were estimated by multiplying the average rainwater concentrations with the annual precipitation (1,300 mm) and the lake area (10.5 km^2).

3.5.3. Objective 3: Spatio-temporal dynamics in environmental conditions and how they drive plankton (zooplankton and phytoplankton) communities in Lake Kanyaboli

The mean abundance of phytoplankton and zooplankton taxa between seasons (dry and wet) and sites were compared using two-way ANOVA, with season and sites as the main factors and a site × season interaction was assessed. Where there were significant site differences, seasonal data were pooled, and one-way ANOVA was tested for spatial differences, and then separation of means was done using Tukey multiple *post hoc*.

The average rank similarities of phytoplankton and zooplankton communities between the wet and dry seasons were compared using a two-way nested analysis of similarities (ANOSIM), with sampling sites nested within seasons. ANOSIM (R-statistic) varies between 0 and 1, with higher values indicating greater differences between the factors. After that, the community composition of phytoplankton and zooplankton in different sites and seasons was visualized using non-metric multidimensional scaling (NMDS) (Clarke & Gorley, 2006), and the differences were confirmed in ANOSIM. The un-transformed abundances data were used to derive dissimilarity matrices based on the Bray–Curtis coefficients (Bray & Curtis, 1957). The magnitude of the associated stress value was used to assess the goodness of fit of the ordination, with a value of <0.2 corresponding to a good ordination (Kashian *et al.*, 2007).

Similarity percentages analysis (SIMPER) was used to determine key phyto- and zooplankton taxa responsible for the differences observed between seasons which were also indicator taxa for changes in water quality. SIMPER is a strictly pairwise analysis between two-factor levels (Clarke & Warwick, 2001); for example, comparisons between wet and dry seasons were made in this study.

To investigate the relationship between the plankton communities and environmental variables, canonical correspondence analysis (CCA) was used. The triplots were used to visualize the output, where the plotted points for taxa could be related to physico-chemical variables that were represented as rays.

3.5.3.1. Analysis of plankton species diversity

Species diversity indices are quantitative measures of the composition of different species in a community and how evenly they are distributed amongst themselves. The plankton abundance data were used to analyze the commonly used diversity indices such as species richness (S), Shannon's diversity (H'), Simpson's diversity (D_1), Simpson's dominance (D_2), and Simpson's evenness (E) (Simpson, 1949).

3.5.4. Objective 4: Temporal changes in fish species composition, catch and fishing effort in Lake Kanyaboli

Descriptive statistics were used to summarise fishery characteristics data from February 2020 to February 2021 and compare the annual fish composition, catch and fishing effort from historical data. Some diversity indices were used to compare the composition of fish data among months and seasons. The indices used included species richness (S), Shannon's diversity (H'), Simpson's diversity (D_1), Simpson's dominance (D_2), Simpson's evenness (E), Fisher’s alpha index and Berger-Parker index (B).

Kruskal-Wallis (H test) was used to compare fish abundance and catch among months on data collected during February 2020 to February 2021 survey. ANOSIM was used to compare average rank similarities of fish abundance and catch between the dry and wet seasons for data from the same study period (February 2020 to February 2021), followed by permutational multivariate analysis of variance (PERMANOVA) to check for significant differences. After that, NNMDS was used to visualise the fish abundance and catch data with respect to sampling sites and seasons. Dissimilarity matrices were derived using un-transformed fish abundances and catch data based on the Bray–Curtis coefficients (Bray & Curtis, 1957). Furthermore, SIMPER was used to determine fish taxa responsible for potential differences between seasons for statistically different ANOSIM results. SIMPER made comparisons between dry and wet seasons (Clarke & Warwick, 2001).

Catch per unit effort (CPUE, kg / kWday) was calculated as a quotient of total catch and total effective fishing effort (kW) following Zeller *et al*. (2021). The annual fish catch was used for secondary data (Lihanda *et al*., 2003; Abila & Othina, 2005; GoK, 2014; Kimani *et al*., 2018), while the summation of fish caught between February 2020 and February 2021 was used for the year 2020. The boat-based effective fishing effort was used for fishing effort because data on number of fishing boats were available for most years (2003, 2005, 2013 and 2020), while assumptions and approximations were made for 2014 and 2015. Similarly, data on the number of days spent fishing was available for 2005 and 2020, while assumptions and approximations were made for 2003, 2013, 2014 and 2015.

Effective fishing effort (kWdays) is the product of nominal fishing effort in kW and the number of days spent fishing per year. However, nominal fishing effort (kW) (boat-based fishing effort) is the product of the engine capacity and the number of boats operating within a fleet segment each year. The length and motorization were used to determine the capacity per fishing vessel (kW) in the fleet (Table 3) (Greer *et al*., 2019). A fleet is the number of vessels with a similar capacity, i.e., in the same length class and motorization category, and utilizing the same or similar fishing gears (Zeller *et al*., 2021). Lake Kanyaboli fishery is predominantly small-scale, using non-motorized vessels below 7 m; hence the fishing boats were considered to have an equivalent capacity of 0.37 kW (Table 3, Greer *et al*., 2019).

Table 3: Engine capacity of non-motorized fishing vessels by length class for small-scale vessels

Length class	Range (m)	Mean length (m)	Capacity non-motorised (kW·boat^{-1})
1	<7.9	4.5	0.37
2	8 – 15.9	11.3	0.75
3	16 – 24.9	20.0	1.12

Data source: Greer *et al*. (2019).

Statistical analyses for all four objectives were performed with Minitab version 17.1 for one-way ANOVA, two-way ANOVA, Kruskal Wallis, Levene's homogeneity test, and *t*-test. In contrast, PAST version 2.17 (Hammer *et al*., 2001) was used for diversity indices.

Furthermore, CCA, SIMPER, ANOSIM and NMDS were performed with R version 3.3.3. (R-Development-Core-Team, 2017) for plankton and fish communities. Furthermore, Microsoft Office Excel (2016) was used to generate graphs and summarise results in tables as well as calculations for trophic status and water and nutrient balance.

CHAPTER FOUR

RESULTS

4.1. Objective 1: Historical changes in the limnological characteristics and trophic status of Lake Kanyaboli from 1981 to 2020

4.1.1. Temporal changes in the water quality of Lake Kanyaboli

The reviewed literature showed that the most studied variables in Lake Kanyaboli are physical and chemical variables, including temperature, pH, Secchi depth, and electrical conductivity (Table 4). In contrast, only one study by Wilfred *et al*. (2005) investigated chlorophyll-*a* in the lake (Table 4). These results show that dissolved oxygen (DO) ranged between 4.7 mg L^{-1} and 10.6 mg L^{-1} over the years, while pH showed less variability fluctuating between 7.05 (2020) and 8.3 (2017). Although electrical conductivity peaked in 1988 (600 µS cm^{-1}), it has also varied over the years, with 2020 recording 424.43 µS cm^{-1}. Furthermore, temperature and Secchi depth also fluctuated within the ranges of 24.50 to 27.98 ^{o}C and 0.28 to 0.76 m over the years, respectively. Again, Chlorophyll-*a* concentration was higher in 2020 (21.46 µg/L) compared to 2005 (18.30 µg/L).

Unfortunately, the temporal comparison of nutrient loading in Lake Kanyaboli was unattainable due to the scarcity of literature. All the reviewed water quality variables (physical, chemical, and biological) varied considerably except pH (Table 4). The results potentially revealed inter- and intra-year seasonal variations linked to rainfall and drought.

Table 4: Summary of water quality in Lake Kanyaboli between 1981 and 2020 as captured in different studies

	References/ Sources of data									
Variable	**Okemwa 1981**	**LBDA 1988**	**Anyona 1997**	**Maithya 1998**	**LVEMP 2002**	**Mutune *et al.* & Wilfred et al. 2005**	**Babu 2015**	**Jalau 2017**	**Present study 2020**	**Limits**
DO (mgL^{-1})	5.4		7.08	**10.6**	6.79	7.4	7.9	**4.7**	10.50 ± 0.48	6^{abc}
pH	7.7		7.47	7.7	**8.28**	7.2		8.3	**7.05 ± 0.18**	$6.5\text{-}9^{ab}$
EC (µScm–^{1})		**600**		349		**287**			424.43 ± 1.88	1500^{b}
Temperature (^{0}C)				**24.5**	25.28	27		25.6	**27.98 ± 0.39**	**-**
Secchi depth (m)	0.75			**0.28**	0.48			0.67	**0.76 ± 0.06**	-
SRP (µg/L)									51.67 ± 9.83	$<100^{c}$
NH_4^+ (µg/L)									983.33± 21.11	$<1000^{cd}$
NO_2^- (µg/L)									210.00 ± 23.66	$<100^{cd}$
NO_3^- (µg/L)									306.67 ± 17.51	$<1000^{cd}$
TP (µg/L)									503.33±109.12	50^{c}
TN (µg/L)									3041.67±280.1	4000^{d}
Chl-*a* (µg/L)						18.30			21.46±9.94	12^{ab}

Aquatic life water quality limits: APHA (2005)^{a}, Rodier *et al*. (2009)^{b}, ANZECC (2000)^{c}, Fried (2003)^{d}.

4.1.2. Trophic status of Lake Kanyaboli

Chlorophyll-*a* concentration results from 2005 and 2020 showed that Lake Kanyaboli is eutrophic, while total phosphorus (2020) indicated hypereutrophic conditions (Table 5). Likewise, Secchi depth in 2020 showed eutrophic conditions in the lake. However, previous studies on Secchi depth (Okemwa, 1981; Maithya, 1998; LVEMP, 2002; Jalau, 2017) have shown that the lake was eutrophic in 1981 and 2017, while in 1998 and 2002, was characterized by hypereutrophic conditions (Table 5). Overall, the high (0.76 m) water clarity was observed in 2020, while 1998 had the lowest (0.28 m) water clarity (Figure 5). The Carlson trophic status index (CTSI) for February 2020 to February 2021 survey averaged 75.82 based on SD, Chl-*a*, and TP, denoting that Lake Kanyaboli was overall eutrophic in 2020. The TN and TP result from February 2020 to February 2021 further yielded a TN (3041.67±280.10 µg/L) to TP (503.33±109.12 µg/L) ratio of 6:1.

Table 5: Trophic status index of Lake Kanyaboli between 1981 and 2020 based on Secchi depth, total phosphorus, and chlorophyll-*a* concentrations.

Reference	**Secchi Depth (m)**	**TSI**	**Trophic Class**
Okemwa, 1981	0.75	64.15	Eutrophic
Maithya, 1998	0.28	78.37	Hypereutrophic
LVEMP, 2002	0.48	70.59	Hypereutrophic
Jalau, 2017	0.67	65.78	Eutrophic
Present study, 2020	0.76	63.96	Eutrophic
	Total phosphorus (μg/L)		
Present study, 2020	500	93.81	Hypereutrophic
	Chlorophyll-*a* (μg/L**)**		
Wilfred *et al*., 2005	18.30	68.12	Eutrophic
Present study, 2020	21.46	69.68	Eutrophic

TSI = Trophic Status Index

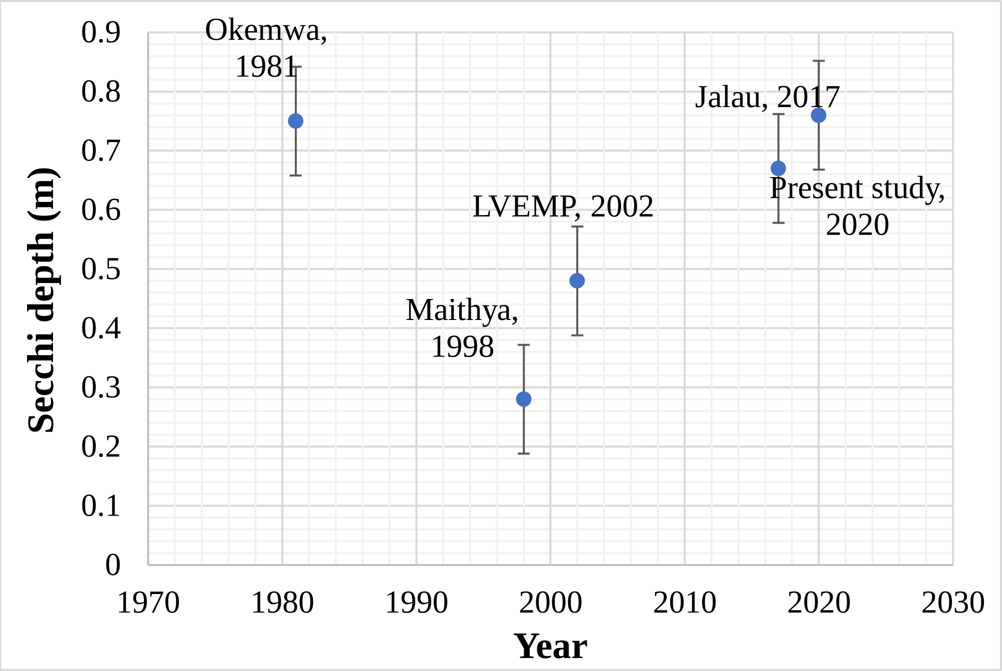

Figure 5: Water transparency in Lake Kanyaboli water transparency over time

4.1.3. Water quality changes in Lake Kanyaboli

The spatial data from six sampling sites on water quality variables which satisfied homogeneity of variance ($p > 0.05$) (Appendix II), were pooled, and summarized using mean for the entire study period (Figure 6). All investigated water quality variables were high in dry seasons except pH, which was high during the wet season (Figure 6). Two-way ANOVA showed no significant interaction between site and season for all water quality variables (Appendix III). However, there were significant seasonal variations in Secchi depth (p = 0.008), dissolved oxygen (p = 0.034), ammonium (p = 0.026), nitrite (p = 0.013), total nitrogen (p = < 0.001) and chlorophyll-*a* (p = 0.002) (Appendix III).

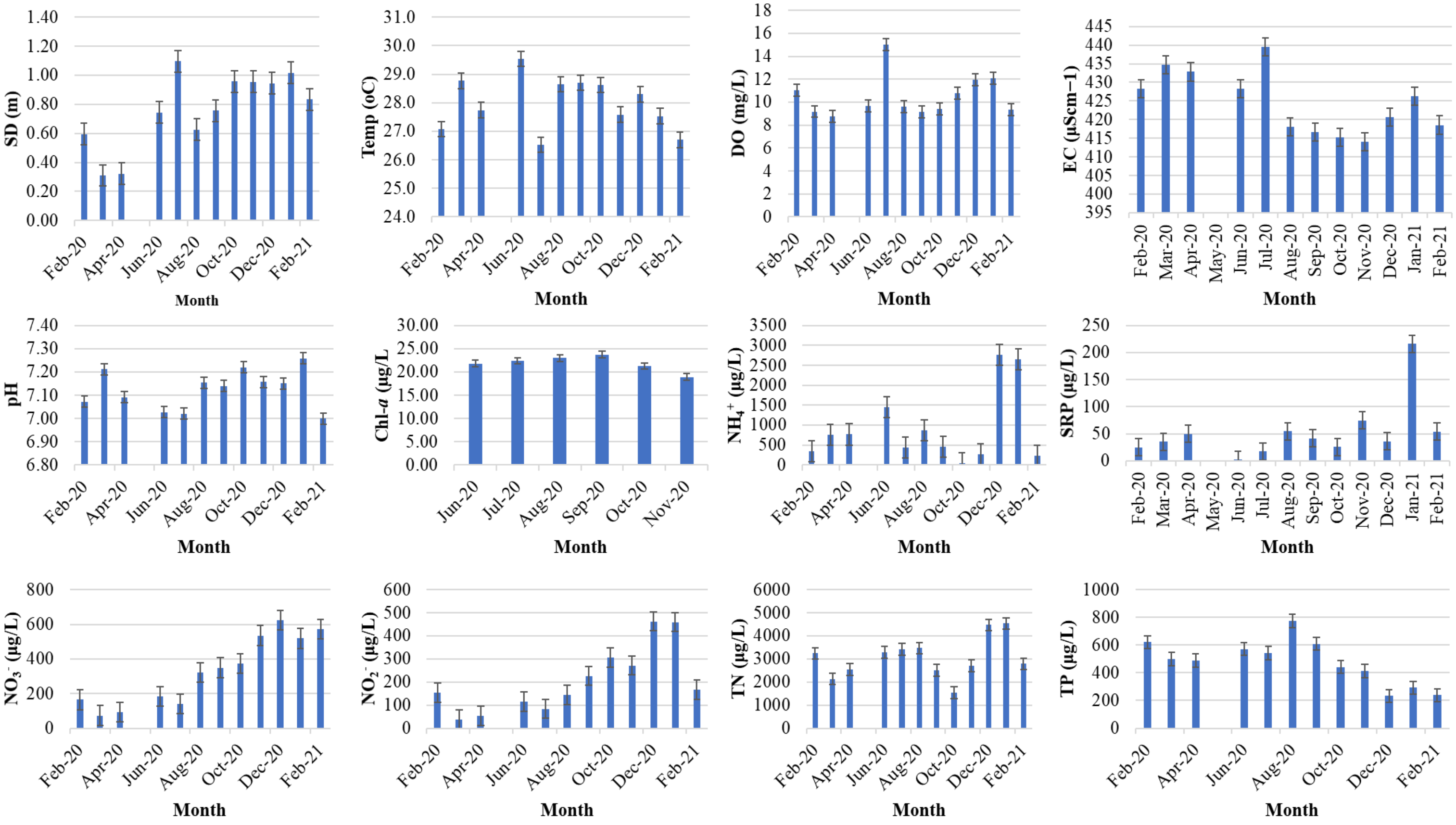

Figure 6: Temporal variation in physical, chemical, and biological variables in Lake Kanyaboli from February 2020 to February 2021. SD: Secchi depth; Temp.: Temperature; EC: Electrical conductivity; DO: Dissolved oxygen; TP: Total phosphorus; TN: Total nitrogen, Chl-*a*: Chlorophyll-*a*, SRP: Soluble reactive phosphorus; NH_4^+: Ammonium; NO_3^-: Nitrate; NO_2^-: Nitrite..

The PCA biplot results showed that physico-chemistry data on sites were not separated, but seasons were (Figure 7a, b). Principal components 1 (PC 1) and 2 (PC 2) of the PCA explained 33.0% and 17.1% of the total variation in water quality variables, respectively. These findings (PCA) suggest a stronger seasonal effect compared to the spatial effect.

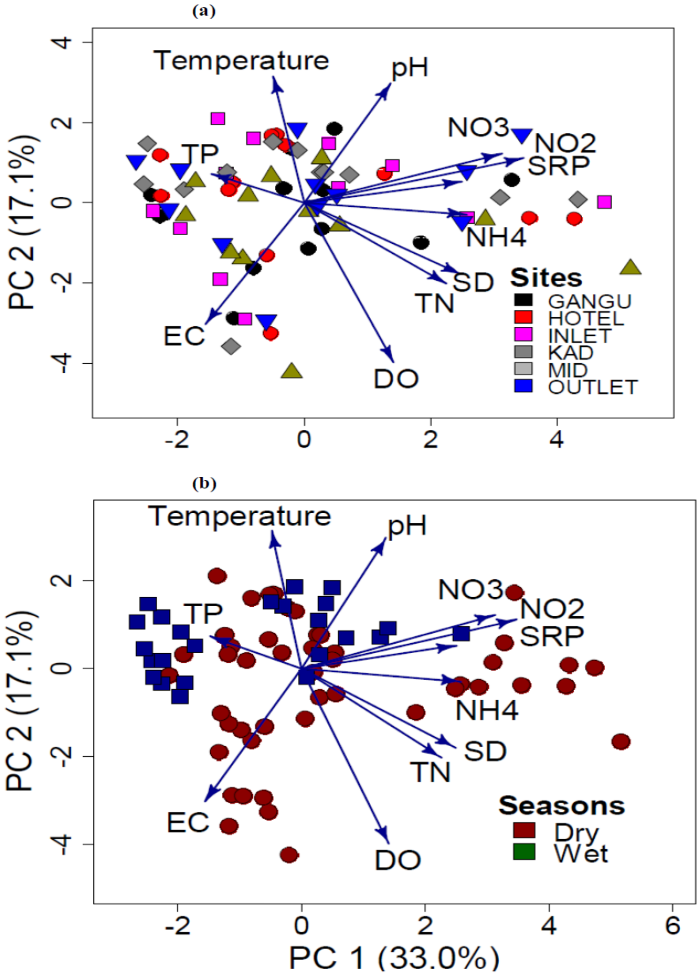

Figure 7: PCA comparison of physico-chemical variables in Lake Kanyaboli based on: (a) site (b) season from February 2020 to February 2021. KAD: Kadenge, MID: Open water

Site-wise mean variations indicated that chlorophyll-*a* varied more among the water quality variables studied (Table 6). Chlorophyll-*a* concentration was lowest (9.03 ± 0.81 μg/l) in Open Water (station 2), while Prince's Hotel (station 3) had the highest chlorophyll-*a* concentration (34.97 ± 3.36 μg/l).

Site comparisons showed no significant variations for physical and chemical variables ($p > 0.05$), while chlorophyll-*a* differed significantly among the sites ($F_{(5, 30)} = 61.54$, $p = <0.001$) (Table 6). Turkey's p*ost hoc* multiple comparisons of means showed that the chlorophyll-*a* concentration in the littoral zone at the Hotel (station 3) (34.97 ± 3.36 μg/L) and Kadenge (station 4) (28.49 ± 3.06 μg/L) sampling sites were significantly different from each other and the rest of the sites while littoral zone site at Gangu (station 1) (23.52 ± 1.11 μg/L) and inflow zone site at Inlet Lake Side (station 5) (21.45 ± 4.19 μg/L) were the same but significantly different from the rest of the sites. In addition, the Open water site (station 2) (9.03 ± 0.81) and outflow zone site at Outlet Lake Side (Kombo, station 6) (11.29 ± 2.03 μg/L) were the same but significantly different from other sites.

Table 6: Mean values (±StdDev) among sampling stations in Lake Kanyaboli, Kenya, from February 2020 to February 2021. Means with different letters across the row are significantly different at α = 0.05 as determined by multiple comparisons (one-way ANOVA followed by post hoc Tukey)

Variable	**Gangu**	**Open Water**	**Hotel**	**Kandenge**	**Inlet-Lake**	**Outlet-Lake**	***p*-value**
SD (m)	0.74 ± 0.30^{a}	0.87 ± 0.34^{a}	0.72 ± 0.27^{a}	0.69 ± 0.29^{a}	0.80 ± 0.24^{a}	0.74 ± 0.25^{a}	0.795
Temp. (^{0}C)	27.63 ± 1.11^{a}	27.49 ± 0.79^{a}	27.86 ± 1.03^{a}	28.26 ± 1.26^{a}	28.14 ± 1.78^{a}	28.50 ± 1.51^{a}	0.452
DO (mg/L)	10.76 ± 2.03^{a}	10.84 ± 2.99^{a}	9.69 ± 2.46^{a}	10.14 ± 2.38^{a}	10.86 ± 2.04^{a}	10.71 ± 2.77^{a}	0.779
EC (μScm^{-1})	423.61 ± 9.83^{a}	427.59 ± 14.70^{a}	423.62 ± 10.57^{a}	423.37 ± 10.69^{a}	422.59 ± 8.20^{a}	425.81 ± 12.99^{a}	0.918
pH	7.12 ± 0.09^{a}	7.13 ± 0.11^{a}	6.68 ± 1.50^{a}	7.14 ± 0.09^{a}	7.11 ± 0.09^{a}	7.14 ± 0.10^{a}	0.433
NH_4^+ (μg/L)	786.22 ± 999.79^{a}	938.13 ± 1143.72^{a}	1134.77 ± 1042.21^{a}	1023.10 ± 1085.10^{a}	1075.82 ± 1023.98^{a}	941.36 ± 839.70^{a}	0.356
SRP (μg/L)	46.09 ± 60.83^{a}	44.45 ± 60.98^{a}	51.03 ± 64.02^{a}	46.99 ± 55.16^{a}	48.07 ± 56.27^{a}	74.45 ± 61.49^{a}	0.613
NO_3^- (μg/L)	331.43 ± 225.70^{a}	279.58 ± 184.06^{a}	317.44 ± 175.06^{a}	297.23 ± 197.77^{a}	307.96 ± 216^{a}	301.01 ± 191.51^{a}	0.992
NO_2^- (μg/L)	168.51 ± 142.43^{a}	200.41 ± 162.21^{a}	241.27 ± 183.66^{a}	220.72 ± 178.41^{a}	221.90 ± 168.96^{a}	205.56 ± 139.78^{a}	0.835
TN (μg/L)	3233.03 ± 1142.68^{a}	3488.47 ± 982.08^{a}	3024.04 ± 1325.53^{a}	2927.87 ± 1265.82^{a}	2696.22 ± 1247.00^{a}	2881.66 ± 1399.40^{a}	0.544
TP (μg/L)	441.88 ± 137.78^{a}	425.22 ± 144.40^{a}	487.08 ± 113.58^{a}	446.57 ± 146^{a}	719.45 ± 683.36^{a}	489.85 ± 94.35^{a}	0.733
Chl-*a* (μg/L)	23.52 ± 1.11$^{\mathbf{c}}$	9.03 ± 0.81$^{\mathbf{d}}$	34.97 ± 3.36$^{\mathbf{a}}$	28.49 ± 3.06$^{\mathbf{b}}$	21.45 ± 4.19$^{\mathbf{c}}$	11.29 ± 2.03$^{\mathbf{d}}$	**0.000**

StdDev: Standard deviation; SD: Secchi depth; Temp.: Temperature; EC: Electrical conductivity; DO: Dissolved oxygen; TP: Total phosphorus; TN: Total nitrogen, Chl-*a*: Chlorophyll-*a*, SRP: Soluble reactive phosphorus; NH_4^+: Ammonium; NO_3^-: Nitrate; NO_2^-: Nitrite

Selected water quality variables showed significant positive correlations (Table 7). These included between SRP and temperature (r = 0.86), NO_3^- and NH_4^+ (r = 0.68), NO_2^- and NH_4^+ (r = 0.80), TN and NH_4^+ (r = 0.85), NO_3^- and NO_2^- (r = 0.65), NO_3^- and TN (r = 0.59), NO_2^- and TN (r = 0.57) and NO_2^- and chlorophyll-*a* (r = 0.57). In contrast, Secchi depth showed significant negative correlation with NH_4^+ (r = -0.77), NO_3^- (r = -0.82), NO_2^- (r = -0.77) and TN (r = -0.64) and chlorophyll-*a* (r = -0.51).

Table 7: Pearson correlation coefficients within various water quality variables in Lake Kanyaboli from February 2020 to February 2021

	SD	**Temp.**	**DO**	**EC**	**pH**	**NH_4^+**	**SRP**	**NO_3^-**	**NO_2^-**	**TN**	**TP**
Temp.	-0.43										
DO	-0.15	-0.13									
EC	0.45	-0.34	0.00								
TDS	0.12	-0.42	0.44	-0.02							
pH	-0.25	0.22	0.21	0.15							
NH_4^+	-0.77*	0.23	0.23	-0.31	0.37						
SRP	-0.43	0.86*	-0.04	-0.21	0.17	0.39					
NO_3^-	-0.82*	0.28	0.11	-0.33	0.37	0.68*	0.41				
NO_2^-	-0.77*	0.38	0.28	-0.36	0.22	0.88*	0.41	0.65*			
TN	-0.64*	-0.02	0.37	-0.04	0.46	0.85*	0.23	0.59*	0.57*		
TP	-0.02	-0.16	-0.17	-0.26	0.04	0.38	0.01	0.23	0.31	0.11	
Chl-*a*	-0.51*	0.12	-0.36	-0.42	-0.30	0.35	0.20	0.57*	0.41	0.17	0.16

*Correlation coefficient is significant at α = 0.05.

4.2. Objective 2: Factors that influence nutrient (nitrogen and phosphorus) dynamics in Lake Kanyaboli using a mass balance model

4.2.1. Water balance

The water balance was modelled based on average rainfall, evaporation, inflow and outflow. The previous study by Odwori & Wakhungu (2021) showed that the average monthly rainfall around Lake Kanyaboli is 90.99 mm, with a peak in April (181.94 mm), while June received the least rain (43.50 mm). The annual rainfall was estimated at 1095 mm year^{-1} (Odwori & Wakhungu, 2021). However, other studies have reported annual rainfall of 1300 mm year^{-1} (Anyona, 1997), 1500 mm year^{-1} (Okungu *et al*., 2005) and 1100 (Mwakubo & Ikiara, 2008). Therefore, annual rainfall of 1300 mm year^{-1} was used in the water balance model for the present study. The evaporation rate was estimated at 1800 mm annually (Akwira *et al*., 2016).

Literature on the inflow into- and outflow from Lake Kanyaboli is lacking; hence the data collected between February 2020 and February 2021 was utilised for the water balance calculation. The monthly rate of discharge into the lake from the influent canal was lower in August 2020 (0.15 m^{3}s^{-1}) and higher in February 2020 (2.57 m^{3}s^{-1}) (Figure 8), with a mean discharge rate of 1.22 m^{3}s^{-1}. On the other hand, outflow discharge was higher (1.81 m^{3}s-1) in August 2020, while February 2020 recorded lower (0.13 m^{3}s^{-1}) outflow discharge (Figure 8). The mean outflow discharge rate between February 2020 and February 2021 was 0.94 m^{3}s^{-1}. Therefore, the mean discharge into the lake was 22.95 % more than the outflow.

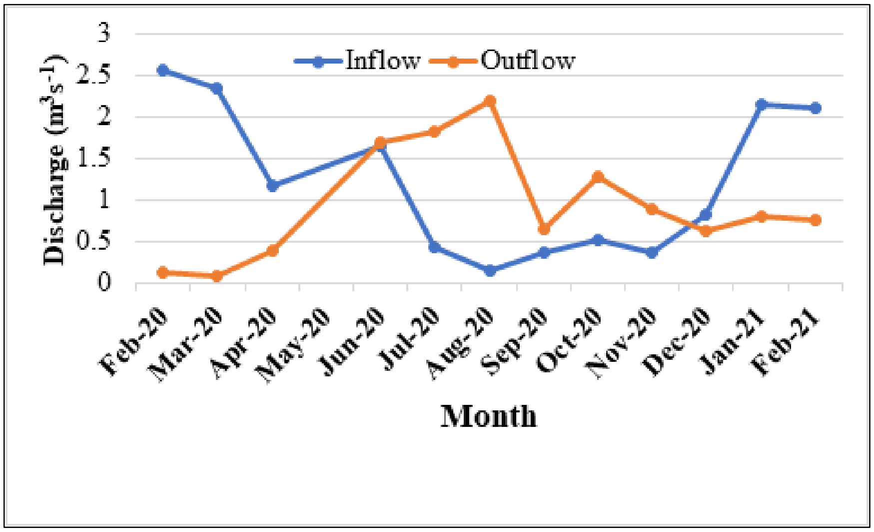

Figure 8: Monthly discharge rates into and out of Lake Kanyaboli from February 2020 to February 2021.

The average depth of the lake during the February 2020 and February 2021 study period was 3.4 m, with wet and dry seasons being 3.53 m and 3.38 m deep, respectively. The lake volume was estimated at 0.036 km^3 during the twelve-month sampling period (Table 8). The estimated volume was 8.75 % higher than the literature-derived estimations (0.0315 km^3) based on the reported surface area (10.5 km^2) and depth (3 m) (Abila *et al.*, 2008; Opiyo and Dadzie, 1994; Muoria *et al.*, 2015). Overall, the inflow (influent canal) contributed 0.028 km^3 annually at an average discharge rate of 1.22 m^3 $s^{-1,}$ while rainfall contributed 0.017 km^3 annually (Table 8). On the other hand, Lake Kanyaboli loses water annually through outflow (outlet canal) (0.026 km^3) and evaporation (0.020 km^3) (Table 8).

However, based on the estimated parameters, the increase in water depth (from 3 to 3.4 m) and volume between February 2020 and February 2021 perhaps shows that groundwater inflow is an essential component of the Lake Kanyaboli water balance. Moreover, the retention dyke was raised during the reinforcement by Dominion Farms Ltd, which could have also increased the holding capacity of the lake (Burgess, 2008).

Table 8: Estimated hydrological parameters of Lake Kanyaboli based on the present study from February 2020 to February 2021.

Variable	**Volume (km^3)**
Lake Volume	0.036
Input	
Inflow	0.028
Rainfall	0.017
Output	
Outflow	0.026
Evaporation	0.020

The water balance estimates indicate that the lake loses (outlet and evaporation) more water than it receives (influent canal and rainfall), and the deficit was 0.001 km^3 (Table 9). However, the resultant increase in depth from 3 m to 3.4 m despite the imbalance could be linked to groundwater inflow, an important water source in wetland lakes (Jolly *et al*., 2008; Schwerdtfeger *et al*., 2014). However, groundwater inflow was not estimated in the current study. The assumption on the importance of groundwater inflow in Lake Kanyaboli is further strengthened by the resilience of the lake water level when it lacked surface inflow between the 1980s and 2004 (Aloo, 2003; Abila, 2005). Furthermore, Lake Kanyaboli is fringed by a think papyrus (*Cyperus papyrus*) macrophytes that are either anchored or free-floating, forming floating islands which further lead to water loss (not measured in this study). Therefore, if input from seepage were negligible, the lake would have shrunk considerably and/or completely dried during dry months when it lacked river inflow. Thus, the resilience in water levels of the lake during the period (the 1980s – 2004) indicates an unaccounted water source from groundwater flow.

The water residence time based on the current mean discharge rate (1.22 ms^{-1}) is 0.80 years (292 days) (Table 9). However, the influent canal has a potential discharge rate of 5 ms^{-1} when operated at total capacity based on an improved design by Dominion Farm Ltd (Burgess, 2008). This means the residence time of the water would drastically reduce to 0.27 years (98.55 days) if the lake volume remains 0.036 km^3.

Table 9: Estimated water budgets for Lake Kanyaboli based on the present study from February 2020 to February 2021.

Variable	Volume (km^3)
Input	0.045
Output	0.046
Net difference	-0.001
Residence time	0.8 years (292 days)

4.2.2. Nutrient balance

The monthly nutrient loads of total nitrogen (N) and phosphorus (P) in water flowing to Lake Kanyaboli and out of the lake showed varying loads between February 2020 and February 2021 (Figure 9). Total nitrogen loads into the lake were more in November 2020 (23.38 t N month^{-1}), while March 2021 had the least load (2.44 t N month^{-1}). The month of June 2020 lost more N loads (25.12 t N month^{-1}) from outflow, with the least load recorded in March 2020 (1.46 t N month^{-1}). Likewise, February 2020 contributed more nutrient load of P (2.45 t P month^{-1}) into the lake compared to December 2020, which had the least P load (0.10 t P month^{-1}) (Figure 9). The P loads transported out of the lake were more in June 2020 (4.54 t P month^{-1}), while December 2020 had the least (0.08 t P month^{-1}).

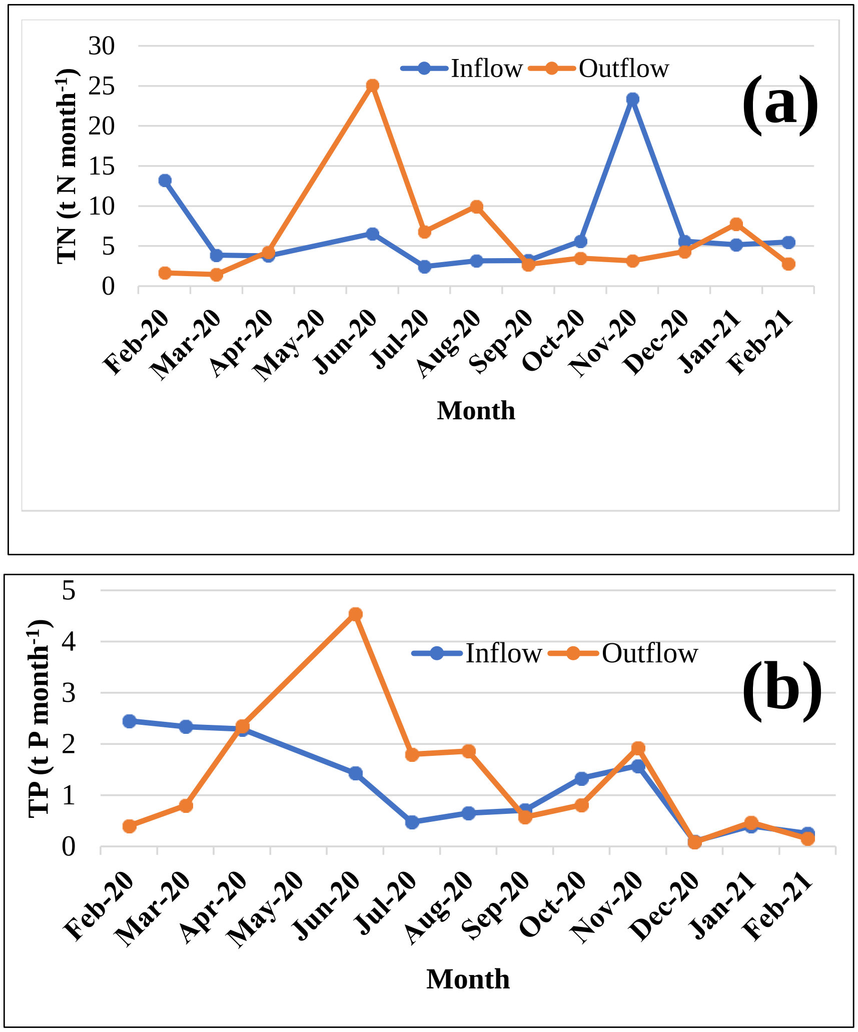

Figure 9: Monthly total nitrogen and phosphorus loads into and out of Lake Kanyaboli between February 2020 and February 2021. (a) = total nitrogen and (b) = total phosphorus.

The nutrient loads for nitrogen (N) in the atmospheric deposition, inflow and outflow showed higher loads in the wet season than dry seasons (Table 10). Likewise, phosphorus (P) loads were high in the wet season for inflow and outflow, while atmospheric deposition had high nutrient loads in the dry season (Table 10).

Table 10: Seasonal variations in Lake Kanyaboli nutrient loads from the atmosphere, inflow and outlet between February 2020 and February 2021.

Nutrients		
TN (t N year^{-1})	**Dry season**	**Wet season**
Input		
Atmospheric deposition	3.4	5.96
River inflow	32.65	48.76
Output		
River outflow	31.66	41.76
TP (t P year^{-1})		
Input		
Atmospheric deposition	6.32	3.75
River inflow	4.93	9.06
Output	11.25	12.81
River outflow	5.25	10.51

The annual nutrient loads into Lake Kanyaboli showed that river inflow (inlet canal) contributed more input for both N (81.42 t N year^{-1}) and P (13.99 t P year^{-1}) (Table 11) mass. Similarly, dissolved fractions of N, including NH_4^+ (37.01 t N year^{-1}) and NO_3^- (23.61 t N year^{-1}), were also more in the inflow than atmospheric deposition (wet and dry) that had 5.96 t N year^{-1} and 3.40 t N year^{-1} respectively. Atmospheric depositions for P were 3.75 t P year^{-1} and 6.32 t P year^{-1} for wet and dry deposition, respectively. The outflow also showed a high annual nutrient load loss for N mass (73.42 t N year^{-1}) and P mass (15.76 t P year^{-1}) (Table 11). The lake lost more phosphorus in the outflow (15.76 t P year^{-1}) than it gained in the inlet canal flow (13.99 t P year^{-1}) (Table 11). The atmospheric deposition showed that wet deposition was the second most important nitrogen source (5.96 t N year^{-1}), while dry deposition contributed more towards phosphorus (6.32 t P year^{-1}) besides inflow.

Table 11: Sources of nutrient loads and nutrient losses in Lake Kanyaboli between February 2020 and February 2021.

Attributes	Nutrients				
	TN ($t\ N\ year^{-1}$)	**NH_4^+ ($t\ N\ year^{-1}$)**	**NO_3^- ($t\ N\ year^{-1}$)**	**TP ($t\ P\ year^{-1}$)**	**SRP ($t\ P\ year^{-1}$)**
Input					
Wet deposition	5.96	3.15	1.26	3.75	1.22
Dry deposition	3.4	1.52	0.83	6.32	2.31
River inlow	81.42	37.01	23.61	13.99	3.38
Output					
River outflow	73.42	38.65	14.94	15.76	3.98

The nutrient mass balance in Lake Kanyaboli showed seasonal differences, with N and P inputs showing higher loads in the wet season compared to the dry season (Table 12). There was more N retention in the wet (23.68 %) than in the dry season. Conversely, more P retention was observed in the dry (53.33%) than in the wet season. This shows that the lake is a strong phosphorus sink during the dry season, while N sedimentation is more in the wet season.

Table 12: Seasonal variations in Lake Kanyaboli nutrient loads from the atmosphere, inlet canal and outlet between February 2020 and February 2021. (a): inlet canal and (b): outlet.

	Nutrients	
TN (t N year^{-1})	**Dry season**	**Wet season**
Input	36.05	54.72
Output	31.66	41.76
Net sink	4.39 (12.18 %)	12.96 (23.68 %)
TP (t P year^{-1})		
Input	11.25	12.81
Output	5.25	10.51
Net sink	6 (53.33 %)	2.3 (17.95 %)

Lake Kanyaboli had an in-lake mass of 95.81 t N year^{-1} for nitrogen and 15.85 t P year^{-1} for phosphorus. The yearly total input for N from inflow (81.42 t N year^{-1}), wet deposition (5.96 t N year^{-1}), and dry deposition (3.4 t N year^{-1}) into the lake was 90.78 t N year^{-1} (Figure 10). In comparison, phosphorus had a total load of 24.07 t P year^{-1} derived from inflow (13.99 t P year^{-1}), wet deposition (3.75 t P year^{-1}) and dry deposition (6.32 t P year^{-1}). Likewise, the lake lost 73.42 t N year^{-1} of N through the outflow, while 15.76 t P year^{-1} of phosphorus was also lost through the outflow (Figure 10). The nutrient retention in the lake was 19.12 % (19.12 t N year^{-1}) for nitrogen and 33.96 % (8.31 t P year^{-1}) for phosphorus (Figure 10).

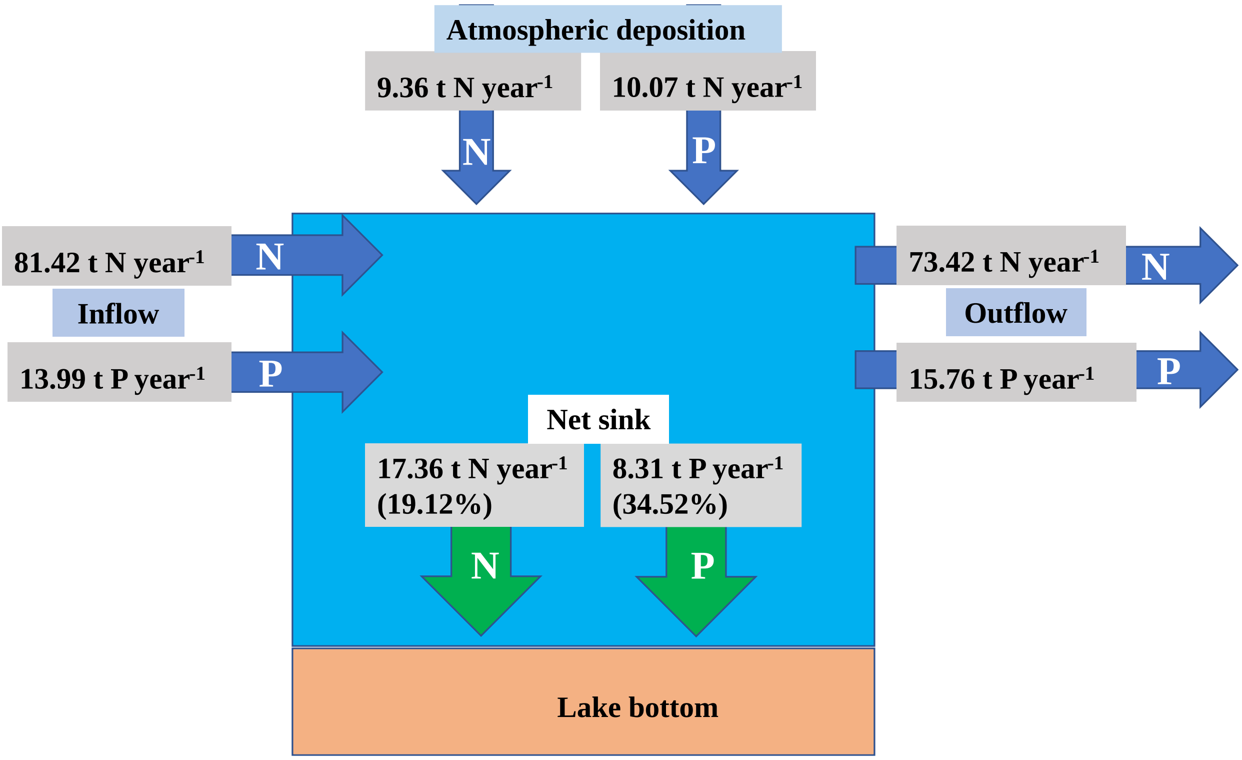

Figure 10: Schematic diagram showing the nutrient sources and losses in Lake Kanyaboli from February 2020 to February 2021.

The mass balance results show that part of the phosphorus (34.5 %) and nitrogen (19.14 %) are buried in the sediments through sedimentation and sequestration. Thus, sediments contribute to internal nutrient loading in the lake besides organic matter and groundwater inflow. At the same time, some nutrients in the sink are lost to diatoms, sunken rotten macrophytes and denitrification. Therefore, assumptions were made that (i) in the absence of inflow nutrient load, the internal loading of nutrients and atmospheric deposition should be able to sustain the primary production of the ecosystem, (ii) although this study did not quantify nutrient load from the excretory products of animal watering in the lake, these are also potentially important sources of nutrients in the lake.

4.3. Objective 3: Spatio-temporal dynamics in environmental conditions and how they drive plankton communities in Lake Kanyaboli

4.3.1. Phytoplankton and zooplankton community composition in Lake Kanyaboli

The phytoplankton community showed variations in the number of taxa recorded between February 2020 and February 2021, with March 2020 having the highest number of taxa (24) while November 2020 had the lowest (6) (Table 13). Likewise, phytoplankton diversity was high in March 2020 based on the Shannon diversity index (H') and the Simpson's diversity index (D_1), while November 2020 had the lowest diversity (Table 13). Simpson's evenness index (E) indicated that October 2020 and February 2021 had the lowest and highest evenness, respectively. Similarly, November 2020 and March 2020 had higher (0.23) and lowest (0.07) Simpson's dominance (D_2), respectively.

Zooplankton also had a varying number of taxa during the study period, with February 2020, March 2020, November 2020, December-2020, and January 2020 having the highest number of taxa (7) while the lowest number of taxa were in July 2020 (2) (Table 13). December-2020 was associated with high Shannon (H') and Simpson's (D_1) diversity indices, while July 2020 had the lowest (Table 13). Although July 2020 had a high (0.74) Simpson's dominance (D_2), the same month had the lowest (0.77) Simpson's evenness index (E) (Table 13).

Table 13: Temporal changes in diversity indices and taxa for phytoplankton and zooplankton communities in Lake Kanyaboli from February 2020 to February 2021

	20-Feb	**20-Mar**	**20-Apr**	**20-Jun**	**20-Jul**	**20-Aug**	**20-Sep**	**20-Oct**	**20-Nov**	**20-Dec**	**21-Jan**	**21-Feb**
Phytoplankton												
Number of taxa	20	24	17	12	10	10	11	11	6	8	8	7
Number of individuals	1070	935	666	807	1057	948	943	675	658	842	642	638
Dominance	0.09	0.07	0.11	0.12	0.15	0.15	0.15	0.16	0.23	0.22	0.17	0.17
Simpson Index	0.91	0.93	0.89	0.88	0.85	0.85	0.85	0.84	0.77	0.78	0.83	0.83
Shannon Diversity Index	2.66	2.89	2.48	2.22	2.02	2.07	2.10	2.02	1.56	1.74	1.84	1.86
Evenness Index	0.71	0.75	0.70	0.77	0.76	0.79	0.74	0.69	0.79	0.71	0.79	0.92
Zooplankton												
Number of taxa	7	7	6	5	2	4	5	6	7	7	7	4
Number of individuals	147	127	62	105	109	137	139	105	88	194	145	120
Dominance	0.20	0.20	0.20	0.22	0.74	0.28	0.22	0.22	0.18	0.16	0.17	0.30
Simpson Index	0.80	0.80	0.80	0.78	0.26	0.72	0.78	0.78	0.82	0.84	0.83	0.70
Shannon Diversity Index	1.78	1.75	1.69	1.55	0.43	1.32	1.56	1.63	1.81	1.88	1.85	1.30
Evenness Index	0.85	0.82	0.90	0.95	0.77	0.94	0.95	0.85	0.89	0.93	0.91	0.91

Chrolophyceae (green algae) was more abundance among the phytoplankton group at all 6 sampling sites (Figure 11a). On the other hand, Bacillariophyceae was the least abundant group in the outflow zone (Outlet lake Side). Likewise, Cryptophyceae had the lowest abundance in the littoral zone (Gangu, Hotel, and Kadenge), inflow zone (Inlet lake Side), and open water stations (Figure 11a). Zooplankton abundance showed that 2 groups dominated the 6 stations: rotifers that were dominant in the inflow zone (Inlet lake Side) and the outflow zone (Outlet lake Side). In contrast, copepods were dominant in the littoral zone (Gangu, Hotel, and Kadenge) and open water (Figure 11b).

Two-way ANOVA showed that there was no significant interaction between site and season in the phytoplankton ($F_{(5, 2148)} = 1.25$, $p = 0.281$) and zooplankton ($F_{(5, 1068)} = 0.71$, $p = 0.618$) abundance. However, the main effects showed that abundances differed significantly between dry and wet seasons for phytoplankton ($F_{(5, 2148)} = 36.61$, $p = <0.001$) and zooplankton ($F_{(5, 1068)} = 101.49$, $p = <0.001$) while no spatial differences were observed for phytoplankton and zooplankton ($p > 0.05$).

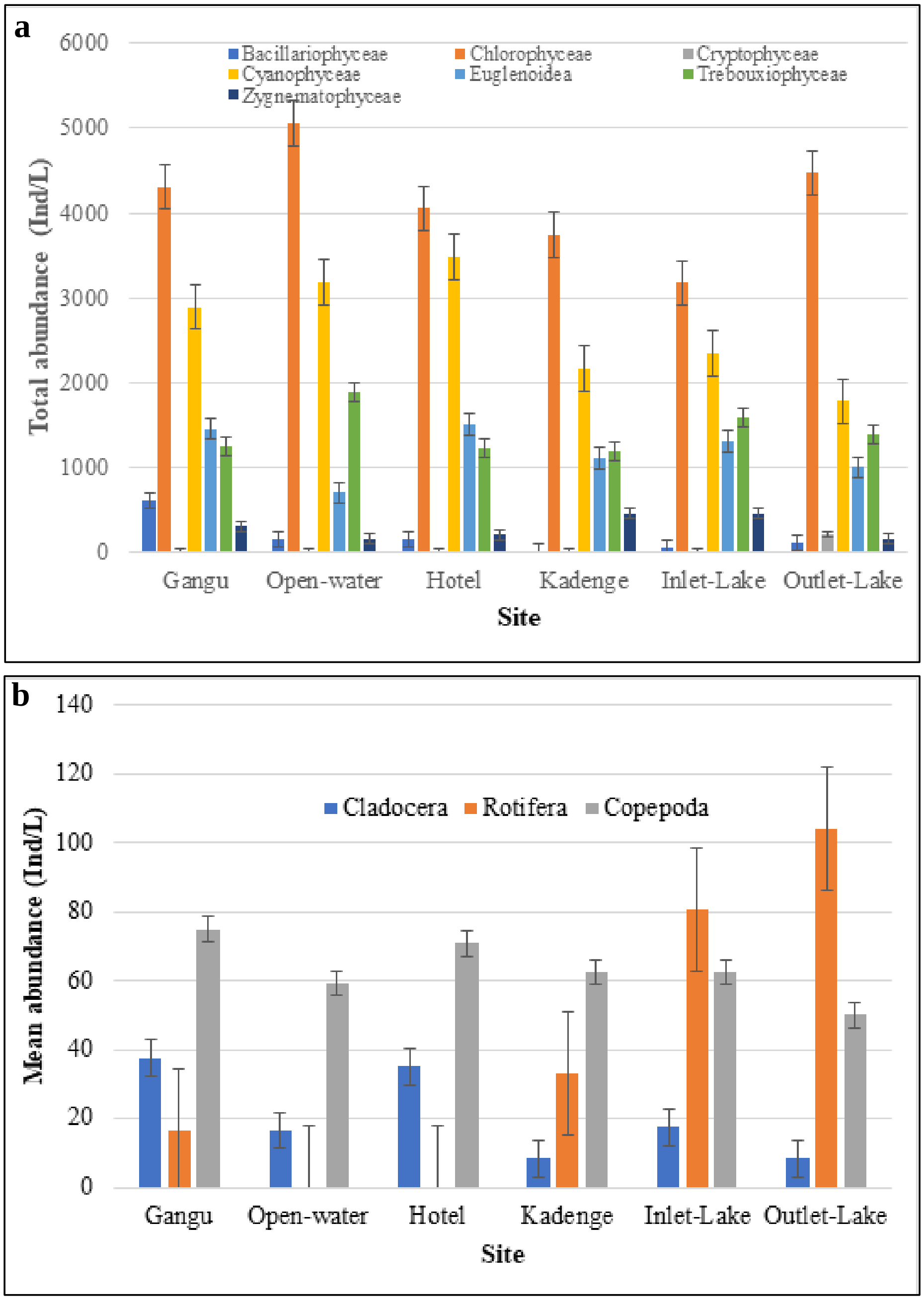

Figure 11: Comparison of mean plankton abundance at different sampling stations in Lake Kanyaboli from February 2020 to February 2021 (a: phytoplankton; b: zooplankton)

January 2020 and February 2021 had the lowest phytoplankton abundance, while February 2020 had the highest abundance (Figure 12a). On the other hand, zooplankton abundance was lowest in April 2020, while December-2020 had the highest abundance (Figure 12b). In general, phytoplankton and zooplankton abundances were higher in the dry season (January, February, July, August and September) than in the wet season (March, April, June, October, November, and December).

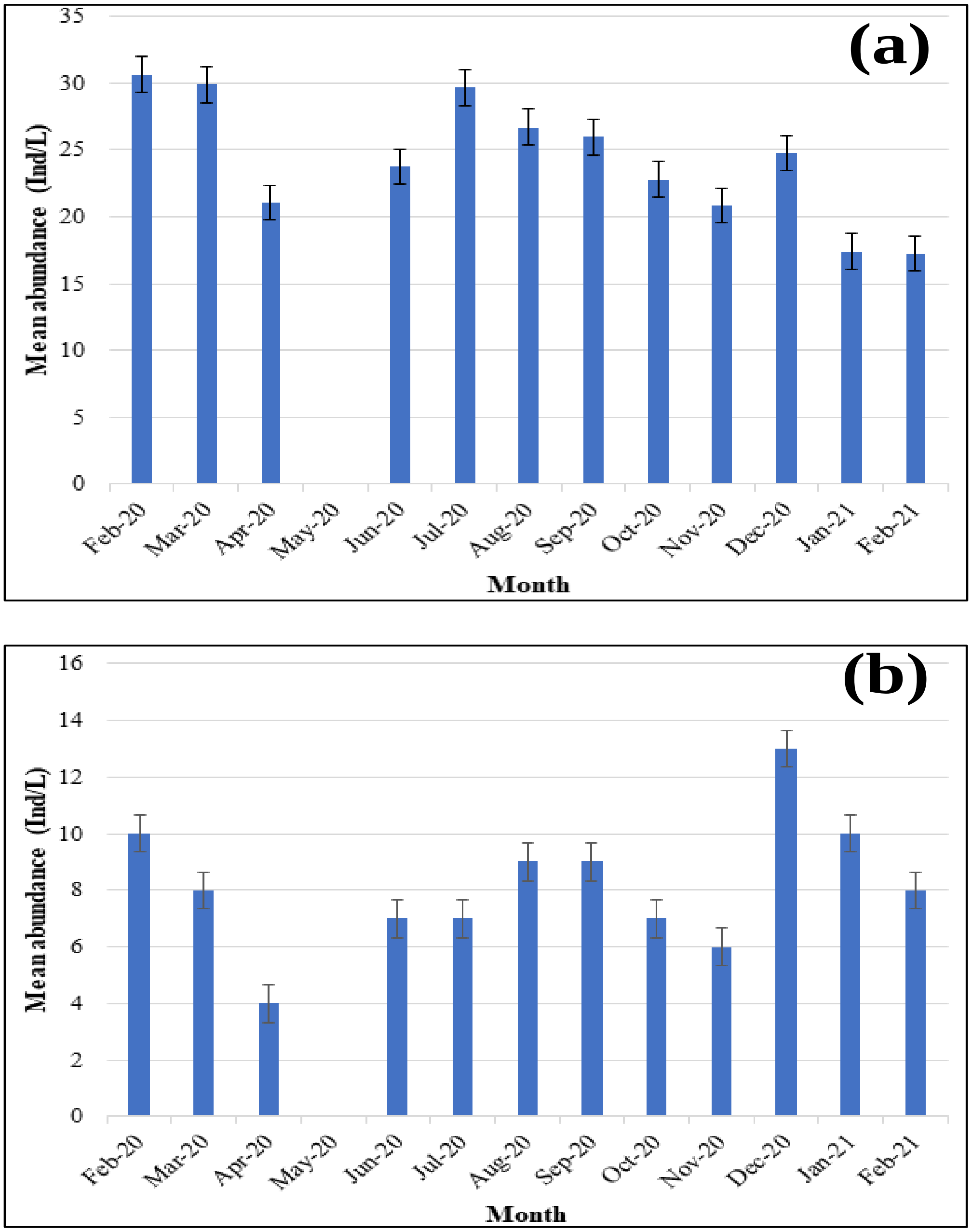

Figure 12: Temporal variations of phytoplankton and zooplankton abundance in Lake Kanyaboli from February 2020 to February 2021 (a: monthly variations of phytoplankton; b: monthly variations of zooplankton).

Phytoplankton had a mean total abundance of 9,913 ± 470 Ind/L in Lake Kanyaboli from February 2020 to February 2021. The phytoplankton community was represented by 30 genera that included Bacillariophyceae (5 genera), Chlorophyceae (9 genera), Cryptophyceae (1 genus), Cyanophyceae (10 genera), Euglenoidae (2 genera), Trebouxiophyceae (2 genera), and Zygnematophyceae (1 genus) (Table 14). These represented broad groups of green algae (Chlorophyceae, Trebouxiophyceae, and Zygnematophyceae), blue-green algae (Cyanophyceae), euglenophytes (Euglenoidae) and diatoms (Bacillariophyceae).

Collectively phytoplankton composition was dominated by green algae (42.3%), followed by blue-green algae (26.93%), euglenophytes (11.85%), and diatoms (1.76%). The relative abundance showed that *Crucigenia* sp. was the most abundant (1,925 Ind/L) taxa, while *Pediastrum* sp. was the least (8 Ind/L) (Table 14). Seasonal comparison of phytoplankton composition revealed that dry and wet seasons recorded the same number of taxa but with varying abundance (Table 14). Three genera were absent in each season, i.e., *Hyalodicus*, *Diploneis* and *Dictyosphaerium* during the wet season and *Diatoma*, *Scenedesmus* and *Pediastrum* during the dry season.

Table 14: Mean relative abundance of phytoplankton taxa recorded in Lake Kanyaboli from February 2020 to February 2021

Class	**Taxon**	**Dry (Ind/L)**	**Wet (Ind/L)**	**Relative contribution (%)**
Bacillariophyceae (1.76%)	*Hyalodicus*	50 ± 18	-	0.5
	Diploneis	42 ± 15	-	0.4
	Navicula	8 ± 3	33 ± 12	0.4
	Diatoma	-	25 ± 4	0.3
	Nitzschia	8 ± 3	8 ± 4	0.2
Chlorophyceae (42.30%)	*Tetraspora*	1016 ± 70	400 ± 102	14.3
	Crucigenia	1408 ± 80	517 ± 55	**19.4**
	Protococcus	392 ± 41	167 ± 14	5.6
	Selenastrum	92 ± 29	75 ± 28	1.7
	Eudorina	25 ± 9	17 ± 5	0.4
	Monoraphidium	17 ± 6	8 ± 4	0.3
	Scenedesmus	-	17 ± 8	0.2
	Pediastrum	-	8 ± 4	**0.1**
	*Microspora_*	17 ± 3	17 ± 5	0.3
Cryptophyceae (0.33%)	*Chilomonas*	-	33 ± 10	0.3
Cyanophyceae (26.60%)	*Merismopedia*	313 ± 65	236 ± 69	5.5
	Polycystic	264 ± 61	83 ± 24	3.5
	Coelosphaerium	208 ± 34	108 ± 43	3.2
	Gomphosphaeria	25 ± 9	142 ± 47	1.7
	Gloeocapsa	158 ± 56	25 ± 13	1.9
	Chrococcus	92 ± 26	58 ± 29	1.5
	Anabaena	58 ± 18	67 ± 20	1.3
	Phormidium	375 ± 15	250 ± 14	6.3
	Aphanocapsa	108 ± 26	42 ± 21	1.5
	Spirulina	8 ± 3	17 ± 5	0.3
Euglenoidea (11.85%)	*Phacus*	667 ± 57	108 ± 20	7.8
	Euglena	333 ± 36	67 ± 12	4.0
Trebouxiophyceae (14.30%)	*Botryoccoccus_*	1008 ± 41	309 ± 14	13.3
	Dictyosphaerium	100 ± 29	-	1.0
Zygnematophyceae (2.86%)	*Closterium*	167 ± 28	117 ± 34	2.9

The abundance of zooplankton averaged 1476 ± 84 Ind/L during the study period. The relative abundance showed that *Nauplius* sp. (266 Ind/L) contributed more towards the total abundance, while *Ceriodaphnia* sp., *Asplanchina* sp. and *Leptodora* sp. contributed the least (17 Ind/L). The present study recorded 15 genera of zooplankton grouped into Rotifera, Copepoda, and Cladocera (Table 15). Copepods were dominant (55.41%), while Cladocerans were the least (16.73%) dominant. Overall, the dry season (14) had more taxa than the wet season (10) (Table 15). *Eubranchipus*, *Ceriodaphnia*, *Diaptomus*, *Microcodon,* and *Asplanchina* were absent during the wet season, while the *Leptodora taxon* was absent during the dry season.

Table 15: Mean relative abundance of zooplankton taxa recorded in Lake Kanyaboli from February 2020 – February 2021

Division	**Class**	**Taxon**	**Dry (Ind/L)**	**Wet (Ind/L)**	**Relative contribution (%)**
Cladocera (16.73%)	Branchiopoda	*Daphnia*	27 ± 9	53 ± 20	5.4
	Branchiopoda	*Diaphanosoma*	70 ± 16	30 ± 4	6.8
	Branchiopoda	*Leptodora*	-	17 ± 8	1.2
	Branchiopoda	*Eubranchipus*	33 ± 8	-	2.2
	Branchiopoda	*Ceriodaphnia*	17 ± 6	-	**1.2**
Copepoda (51.42%)	Hexanauplia	*Nauplius*	233 ± 30	33 ± 6	**18.0**
	Hexanauplia	*Canthocamptus*	98 ± 18	93 ± 8	12.9
	Hexanauplia	*Cyclop*	175 ± 18	60 ± 12	15.9
	Hexanauplia	*Diaptomus*	67 ± 10	-	4.5
Rotifera (31.84%)	Eurotatoria	*Brachionus*	158 ± 20	50 ± 9	14.1
	Eurotatoria	*Hexarthra*	67 ± 15	14 ± 7	5.5
	Eurotatoria	*Kerattela*	50 ± 12	17 ± 5	4.5
	Eurotatoria	*Filina*	50 ± 9	14 ± 5	4.3
	Eurotatoria	*Microcodon*	33 ± 6	-	2.2
	Eurotatoria	*Asplanchina*	17 ± 6	-	**1.2**

ANOSIM results indicated no significant differences in phytoplankton community composition for un-transformed abundance data among sites (R-statistic = 0.01, $p = 0.44$) and between seasons (R-statistic = 0.01, $p = 0.40$). On the other hand, ANOSIM showed significant differences for un-transformed abundance data between seasons (R-statistic = 0.12, $p < 0.0001$) but not among sites (R-statistic = 0.08, $p < 0.89$) for zooplankton. The findings suggest a stronger seasonal effect compared to the spatial impacts for zooplankton.

However, non-metric multidimensional scaling (NMDS) did not show a clear temporal and spatial separation of phytoplankton and zooplankton (Figure 13a-d).

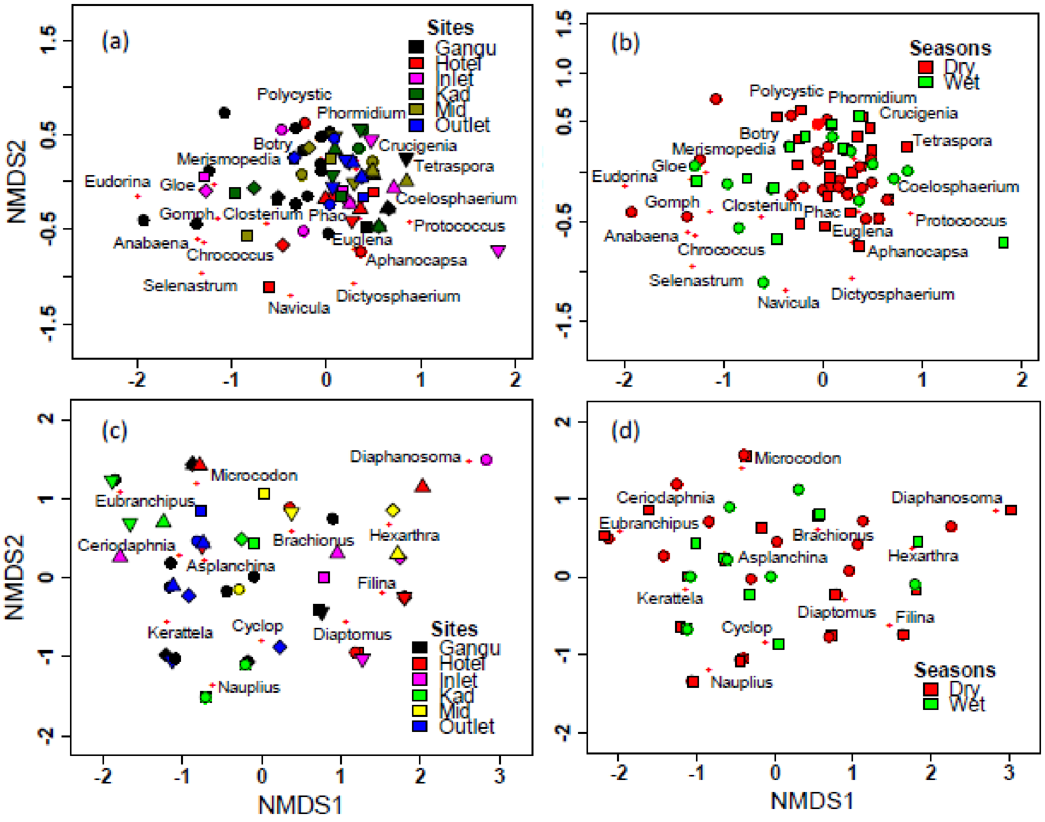

Figure 13: NMDS (stress value = 0.1) comparison of phytoplankton and zooplankton structure in Lake Kanyaboli from February 2020 to February 2021. (a) phytoplankton site comparisons, (b) phytoplankton seasonal comparisons, (c) zooplankton site comparisons (d) zooplankton seasonal comparisons. KAD: Kadenge, MID: Open water

The results of SIMPER analysis for zooplankton showed that *Nauplius* sp. accounted for the most proportion (16.84 %) of zooplankton dissimilarity, while *Ceriodaphnia* sp. accounted for the least (1.31 %) (Table 16).

Table 16: Zooplankton ranked abundance SIMPER contributors to % dissimilarity in phytoplankton composition between the wet and dry seasons.

	Mean		
Taxon	**Dry**	**Wet**	**Contrib. %**
Nauplius	8.3	29.1	16.8
Canthocamptus	23.3	12.3	12.0
Brachionus	12.5	19.9	11.6
Cyclop	15.0	21.8	11.0
Daphnia	13.5	3.4	8.8
Diaphanosoma	7.5	8.8	7.7
Hexarthra	3.5	8.3	6.3
Kerattela	4.3	6.3	5.0
Diaptomus	-	8.4	5.0
Filina	3.5	6.4	4.8
Leptodora	4.3	-	2.8
Eubranchipus	-	4.3	2.8
Microcodon	-	4.1	2.4
Asplanchina	-	2.1	1.6
Ceriodaphnia	-	2.1	1.3

4.3.2. Influence of physico-chemical variables on plankton communities

There were associations between physico-chemical variables and phytoplankton and zooplankton communities. For phytoplankton, the first CCA axis (CCA 1) accounted for the greatest variance (% explained 30.4%-37.0%), while the second axis (CCA 2) accounted for 16.2%-22.8% (Figure 14a, b). Increasing EC correlated positively with phytoplankton taxa such as *Gomphospheria* sp., *Selenastrum* sp., *Chrococcus* sp., *Navicula* sp., and *Anabaena* sp. (Figure 14b). Similarly, *Coelosphaerium* sp. and *Spirulina* sp. were positively correlated with pH, NO_2^{-}, and Secchi depth (Figure 14b). Likewise, *Protococcus* sp., *Botryococcus* sp., *Phormidium* sp., *Crucigena* sp., *Tetraspora* sp., *Eudorina* sp., and *Diatoma* sp. were positively correlated with DO, NO_3^- and SRP (Figure 14b). Furthermore, *Chilomonas* sp., *Phacus* sp., *Polycystic* sp., *Euglena* sp., *Merismopedia* sp., and *Closterium* sp. were negatively correlated with NH_4^+, temperature, TN, and TP (Figure 14b).

For zooplankton, the first CCA axis (CCA 1) also accounted for the greatest variance (% explained 26.2%-31.6%), while the second axis (CCA 2) accounted for 21.7%-29.2% (Figure 14c, d). Zooplankton genera such as *Asplanchina* sp., *Microcodon* sp., *Ceriodaphnia* sp., and *Cyclop* sp. were positively correlated with DO and TN (Figure 14d). At the same time, *Kerattela* sp. was also positively correlated with NO_3^- and Secchi depth (Figure 14d). On the other hand, *Branchionus* sp. showed a negative correlation with TP, NH_4^+, SRP, and EC, while *Hexarthra* sp. and *Nauplius* sp. were positively correlated with temperature (Figure 14d).

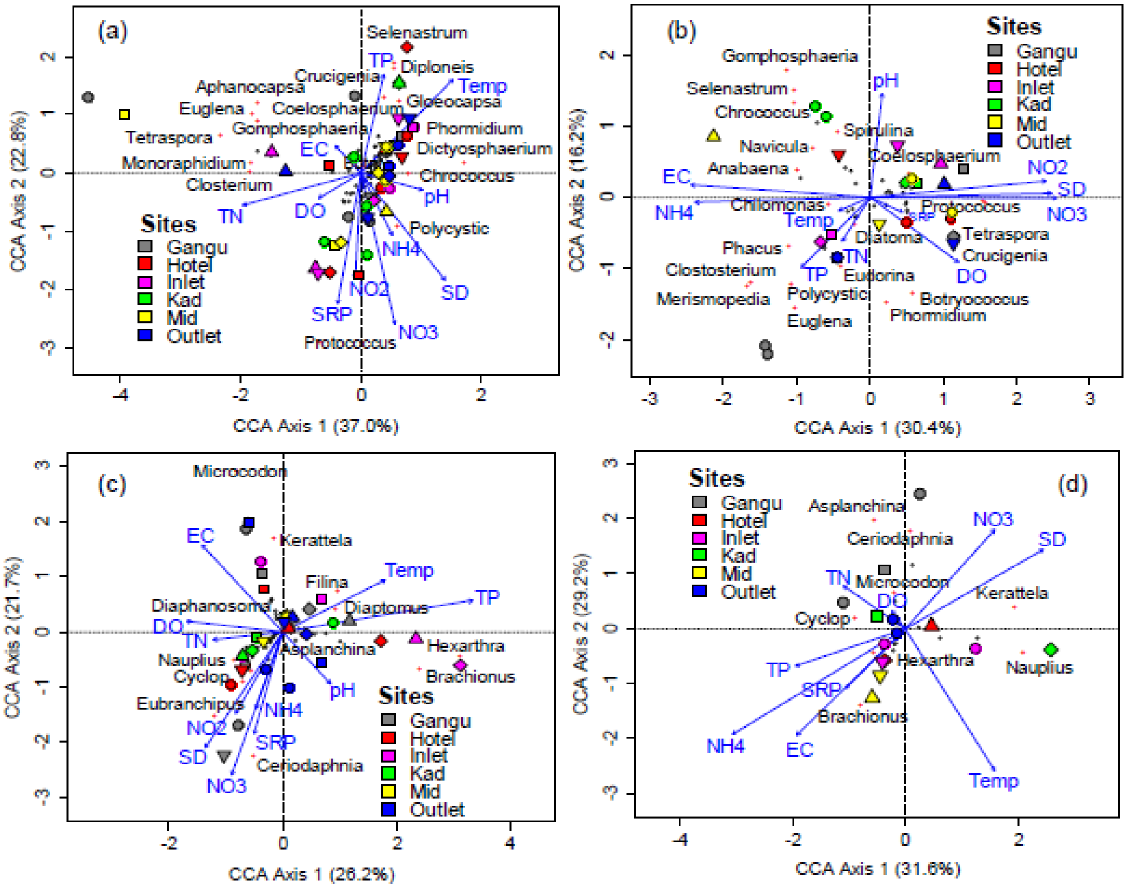

Figure 14: Canonical correspondence analysis (CCA) between plankton and physico-chemical variable among sites in Lake Kanyaboli from February 2020 to February 2021. (a-b) phytoplankton site comparisons, (c-d) zooplankton site comparisons. KAD: Kadenge, MID: Open water

4.4. Objective 4: Temporal changes in fish species composition, catch and fishing effort in Lake Kanyaboli

4.4.1. Characteristics of Lake Kanyaboli fishery

Although the lake was considered a single unit because there were no boundaries on fishing grounds, February 2020 – February 2021 survey collected fishing effort from all three main landing sites (Kadenge, Gamba and Kombo) and two other fishing villages (Hawinga and Swila). Besides the lack of fishing ground demarcation, fishers also land on any landing sites of their choice to sell their catch.

There were 148 fishing boats in Lake Kanyaboli that spent 266 days fishing in a year during February 2020 to February 2021 survey. The days spent fishing represented 73 % of the 365 days a year. Abila & Othina (2005) noted that, on average, fishermen in Lake Kanyaboli would be expected to fish for about 70% (256 days) of the 365 days in a year due to any planned or unexpected disruption. Thus, an average of 261 fishing days (72 % of the number of days in a year) was used to estimate the fishing effort for the years (2005, 2013, 2014 and 2015) with missing data.

The characteristics of fishing effort between 2003 and 2020 showed that 2003 had the lowest number (56) of fishing vessels while 2020 had the highest (148) (Table 17). However, two years (2014 and 2015) had missing data on the number of fishing boats. Therefore, an average number of boats added to the fishery was derived using fishing boat data from four years (2003, 2005, 2013, 2020). The results showed that an average of four fishing boats were added to the fishery each year, translating into 103 and 107 boats for 2013 and 2014, respectively (Table 17). Lake Kanyaboli fishery is predominantly artisanal, targeting tilapia species, haplochromines, small-sized cyprinids, catfish and mudfish. The results from February 2020 to February 2021 survey showed that 390 fishers were operating in Lake Kanyaboli. The mainly used fishing gears are gillnets (with 1, 2, 4 and 5 inch mesh sizes), longline and fish traps.

Table 17: Summary of characteristics of the fishing effort of Lake Kanyaboli between 2003 and 202.

Year	Number of fishing boats	Number of fishers
2003	56[a]	130[a]
2005	65[b]	-
2013	99[c]	188[c]
2014	103	-
2015	107	-
2020	148[d]	390[d]

Note: a = Lihanda *et al*. (2003), b = Abila & Othina (2005), c = Government of Kenya (GoK, 2014), d = Present study (2020), years without letters = derived result using available literature.

4.4.2. Historical trends in fish catches, effort and composition of Lake Kanyaboli

The summary of annual fish catches between 2003 and 2020 from literature, and 2020 survey in Lake Kanyaboli showed variations in the total catch (Table 18). The highest fish catch was recorded in 2003 (262 mt), while 2020 had the lowest landed fish catch (93 mt). The historical data showed that fish catches decreased by 24 % between 2003 (250 mt) and 2005 (199 mt), while there was a similar amount of decrease (25 %) between 2014 (134 mt) and 2015 (100 mt). On the other hand, the fish catch between 2005 (199 mt) and 2013 (188 mt) showed a decrease of 5 %, while the year 2020 (93 mt) indicated a 7 % decrease compared to the amount recorded in 2015.

Effective fishing effort has steadily increased between 2003 and 2020 from 5408 kWdays in 2003 to 14566 kWdays in 2020 (Table 18), representing a 63 % increase in 17 years. The increase in the effective fishing effort had a corresponding decrease in a catch over the same period, from a maximum catch of 262 mt in 2003 to a minimum catch of 93 mt in 2020 (Table 18). Likewise, the catch per unit effort decreased from 47 kg / kWday in 2003 to 6 kg / kWday in 2020 (Table 18).

Table 18: Annual effective fishing effort, catch and catch per unit effort between 2003 and 2020 in Lake Kanyaboli

Year	**Catch (kg)**	**Effective Fishing Effort (kWdays)**	**CPUE (**kg / kWday**)**
2003	261684	5408	48
2005	198863	6157	32
2013	194000	9560	20
2014	134000	9947	13
2015	100000	10333	10
2020	92746	14566	6

Source of catch data: Lihanda *et al*. (2003), Abila & Othina (2005), Government of Kenya (GoK, 2014), Kimani *et al*. (2018) and Present study (2020).

The fish species composition of Lake Kanyaboli has shown great variability over the years, recording between 9 and 15 taxa (Table 19). The comparison between 1981 and 2020 indicated *Oreochromis esculentus*, Haplochromines (particularly *Pseudocranilabrus multicolour victoriae*, *Lipochromis maxillaris*, *Astatoreochromis alluaudi*, *Haplochromis*

(Astatotilapia) nubilus and *Xystichromis phytophagus*), *O. niloticus*, *Protopterus aethiopicus* and *Clarias* sp. fish species were present in all the years considered. The fish species, including *Enteromius paludinosus*, *E. karstenii*, and *E. apleurogramma,* were only reported in 2005, while *Coptodon zillii* was only in 1981 (Table 19). Four recorded fish species are critically endangered, including *Oreochromis esculentus*, *Haplochromis martinii*, *O. variabilis,* and *Xenoclarias* sp. (Table 19).

Table 19: Comparison of fish species between 1981 and 2020 in Lake Kanyaboli

Family	Fish species	Okemwa, 1981	Maithya, 1998	Aloo, 2003	Masai *et al.*, 2005	Present study	IUCN Redlist Status
Cichlidae	*Oreochromis esculentus*	+	+	+	+	+	CE
	Pseudocranilabrus multicolor victoriae	*	-	+	+	+	DD
	Lipochromis maxillaris	*	+	+	+	+	V
	Astatoreochromis alluaudi	*	+	+	+	+	LC
	Haplochromis (Astatotilapia) nubilus	*	-	+	+	+	V
	Astatotilapia sp. bigeye	*	-	+	-	+	-
	Xystichromis phytophagus	*	+	+	+	+	DD
	Haplochromis martini	*	+	-	-	-	CE
	Oreochromis variabilis	+	-	+	-	+	CE
	Oreochromis niloticus	+	+	+	+	+	NE
	Oreochromis leucostictus	+	-	+	-	+	LC
	Coptodon zillii	+	-	-	-	-	LC
Protopteridae	*Protopterus aethiopicus*	+	+	+	+	+	LC
Clariidae	*Clarias* sp.	+	+	+	+	+	LC
	Xenoclarias sp.	+	-	+	-	-	CE
Schilbeidae	*schilbe* sp.	-	-	+	-	-	LC
Bagridae	*Bagrus* sp.	-	-	-	-	+	LC
Anabantidae	*Ctenopoma muriei*	-	-	-	+	+	LC
Poeciliidae	*Aplocheilichthys pumilis*	-	+	+	-	-	LC
Cyprinidae	*Enteromius paludinosus*	-	-	-	-	*	CE
	Enteromius karstenii	-	-	-	+	*	NE
	Enteromius apleurogramma	-	-	-	+	*	NE

Note: "+" means recorded during the study year; "-"means not recorded during the study year. "*" Several Haprochromines and *Enteromius* sp. (formerly Barbus, FishBase (Froese & Pauly, 2018) lumped together. NE = Not evaluated, DD = Data deficient, V = Vulnerable, LC = Least concern, CE = Critically endangered.

4.4.3. Temporal variations in fish composition and catches in Lake Kanyaboli from February 2020 to February 2021

The fish species recorded between February 2020 and February 2021 showed that March 2020 had a high number of individuals (30,577), while January 2021 had the lowest number of individuals (12,770) (Table 20). Eight months (from February 2020 to September 2020 and February 2021) had 14 species each, while a lower number of species was recorded in January 2021 (12). Shannon diversity (*H'*) revealed higher diversity in November 2020 (2.09), while April 2020 had a lower index (1.93). Simpson diversity index ranged between 0.81 in April 2020 (lowest) and 0.84 (highest) in February 2020, March 2020, July 2020, August 2020, November 2020 and January 2020. Likewise, Dominance was high in April 2020 (0.19), while February 2020, March 2020, July 2020, August 2020, November 2020 and January 2021 had the lowest (0.16). In contrast, January 2021 (0.67) fish count showed more evenness, while April 2020 had the least evenness (0.49) (Table 20). Fisher's alpha diversity index was lower in January 2021 (1.27) and higher in September 2020 (1.44). Berger-Parker index ranged between 0.21 and 0.34 representing March 2020 and September 2020, respectively. The dry and wet seasons recorded the same average number of species (14). However, all diversity indices (Simpson diversity index, evenness indices, Fisher-alpha index and Berger-Parker index) were high in the dry season except for dominance and Shannon diversity index (Table 20).

Table 20: Temporal changes in diversity indices and the number of fish taxa in Lake Kanyaboli from February 2020 to February 2021.

	Feb 2020	Mar 2020	Apr 2020	Jun 2020	Jul 2020	Aug 2020	Sep 2020	Oct 2020	Nov 2020	Dec 2020	Jan 2021	Feb 2021
Number of taxa	14	14	14	14	14	14	14	13	13	13	12	14
Number of individuals	24694	30577	26331	21154	15960	20889	14052	15534	13995	15969	12770	14763
Dominance index	0.16	0.16	0.19	0.17	0.16	0.16	0.18	0.17	0.16	0.17	0.16	0.17
Simpson diversity index	0.84	0.84	0.81	0.83	0.84	0.84	0.82	0.83	0.84	0.83	0.84	0.83
Shannon diversity index	2.05	2.03	1.93	1.97	2.06	2.05	2.05	1.96	2.09	2.07	2.08	2.04
Evenness index	0.56	0.54	0.49	0.51	0.56	0.56	0.56	0.55	0.62	0.61	0.66	0.55
Fisher‘s alpha index	1.44	1.40	1.43	1.46	1.51	1.46	1.54	1.40	1.41	1.39	1.31	1.53
Berger-Parker index	0.24	0.21	0.32	0.26	0.26	0.28	0.34	0.29	0.30	0.32	0.28	0.30

A total of 226,686 individuals belonging to 15 taxa were recorded in the lake during the year-long study (February 2020 to February 2021). Interestingly, only one individual of *Bagrus* sp. was registered throughout the study period and hence not included in the summary tables for abundance (Table 21) and catch. Family Cichlidae had more species (10), while Protopteridae, Clariidae, Anabantidae, and Cyprinidae had one species each. *Xystichromis phytophagus* was dominant in the samples (26.31%), while *Ctenopoma muriei* contributed the least number of individuals (0.13%). Collectively Haplochromine species were more abundant and contributed 58.06 % towards total abundance.

Overall, more fish samples were recorded during the wet season (122,275 ± 10186) compared to the dry season (104,410 ± 8211) (Table 22). However, *Clarias* sp., *Enteromius* sp., *Ctenopoma muriei*, *O. leucosticus*, and two *Haplochromis* spp. (*H. (Astatotilapia) nubilus* and *Pseudocranilabrus multicolour victoriae*) were dominant during the dry season, while the rest of the fish species (*O. esculentus*, *O. variabilis*, *O. niloticus*, *Astatotilapia* sp. "bigeye", *Astatoreochromis alluaudi, Lipochromis maxillaris*, *Xystichromis phytophagus* and *Protopterus aethiopicus*) were dominant during the wet season (Table 22). But, all 14 fish species (excluding *Bagrus* sp.) were recorded during dry and wet seasons. Kruskal Wallis H test and *t*-test showed that fish abundance did not vary significantly between months ($H_{(11)} = 2.80$, $p = 0.993$) and seasons ($t_{(136)} = -1.11$, $p = 0.276$), respectively.

Table 21: Species-wise monthly fish abundance caught in Lake Kanyaboli from February 2020 to February 2021

Family	**Fish species**	**Feb 2020**	**Mar 2020**	**Apr 2020**	**Jun 2020**	**Jul 2020**	**Aug 2020**	**Sep 2020**	**Oct 2020**	**Nov 2020**	**Dec 2020**	**Jan 2021**	**Feb 2021**
Cichlidae	*Astatotilapia sp. bigeye*	724	1224	836	612	612	2180	836	1732	1284	1508	1060	948
	Pseudocranilabrus multicolor victoriae	109	137	53	22	24	17	71	33	42	60	112	98
	Xystichromis phytophagus	4192	6483	8509	5489	3565	5920	4778	4527	3542	5151	4158	4473
	Lipochromis maxillaris	1948	6243	3324	2532	1894	2271	1370	1824	924	1432	1455	1516
	Astatoreochromis alluaudi	4558	3148	3294	3102	1684	1324	985	1093	739	969	908	839
	Haplochromis (Astatotilapia) nubilus	631	616	531	554	462	662	631	631	569	708	623	689
	Oreochromis esculentus	5983	6027	4795	4889	4204	3996	1925	2818	1486	1724	1447	2148
	Oreochromis variabilis	56	112	32	41	55	30	0	13	66	0	28	64
	Oreochromis niloticus	3526	4208	3067	2331	1679	2498	1864	2231	2634	2589	2468	2740
	Oreochromis leucostictus	405	689	284	244	325	696	255	260	213	509	505	429
Protopteridae	*Protopterus aethiopicus*	693	861	924	553	616	427	413	119	616	567	490	182
Clariidae	*Clarias* sp.	1771	770	616	693	693	693	770	154	616	616	770	539
Anabantidae	*Ctenopoma muriei*	21	24	31	22	63	35	56	21	0	0	0	14
Cyprinidae	*Enteromius* sp.	77	35	35	70	84	140	70	91	35	70	56	84

Table 22: Relative abundance (percentage, %) contribution of each fish species to the total annual abundance of all fishes caught in Lake Kanyaboli from February 2020 to February 2021.

Family	Fish species	Dry	Wet	Contribution (%)
Cichlidae	*Astatotilapia sp. bigeye*	6360 ± 571	7196 ± 416	5.99
	Pseudocranilabrus multicolor victoriae	431 ± 47	314 ± 34	0.14
	Xystichromis phytophagus	27086 ± 979	33701 ± 1723	26.87
	Lipochromis maxillaris	10454 ± 351	16279 ± 1923	11.82
	Astatoreochromis alluaudi	10298 ± 1427	12345 ± 1238	10.01
	Haplochromis (Astatotilapia) nubilus	3698 ± 80	3609 ± 64	3.23
	Oreochromis esculentus	19703 ± 1741	21739 ± 1875	18.32
	Oreochromis variabilis	233 ± 24	264 ± 43	0.21
	Oreochromis niloticus	14775 ± 661	17060 ± 729	14.07
	Oreochromis leucostictus	2615 ± 154	2199 ± 190	2.13
Protopteridae	*Protopterus aethiopicus*	2821 ± 179	3640 ± 286	2.86
Clariidae	*Clarias* sp.	5236 ± 448	3465 ± 216	3.85
Anabantidae	*Ctenopoma muriei*	189 ± 25	98 ± 13	0.13
Cyprinidae	*Enteromius* sp.	511 ± 29	366 ± 24	0.37
Total		104410	122275	**100.00**

The ANOSIM for fish abundance showed no significant differences between the wet and dry seasons (R-statistic = 0.06, *p* = 0.2518). Permutational multivariate analysis of variance (PERMANOVA) did not also show significant differences between the seasons (F = 1.492, *p* = 0.2176).

The summary of species-wise fish catches in 2020 between February 2020 and February 2021 indicated that *Oreochromis niloticus* contributed more (27.45 %) towards total catch while *Ctenopoma muriei* contributed the least (<1 %) (Table 23). The pulled group data revealed that the main species in catches were Tilapia (50.40 %) followed by *Clarias* sp. (22.45 %), *Protopterus aethiopicus* (19.86%), Haplochromines (7.25 %) (Table 23), but other species comprised of *Ctenopoma muriei* and *Enteromius* sp. (<1 %) were also present.

Table 23: Percentage contribution of each fish species to the total annual catch from Lake Kanyaboli between February 2020 and February 2021

Family	Fish species	Catch (mt)	Percentage Contribution (%)
Cichlidae	*Astatotilapia sp. bigeye*	0.46	0.49
	Pseudocranilabrus multicolor victoriae	0.03	0.03
	Xystichromis phytophagus	3.19	3.44
	Lipochromis maxillaris	1.66	1.80
	Astatoreochromis alluaudi	1.13	1.22
	Haplochromis (Astatotilapia) nubilus	0.25	0.27
	Oreochromis esculentus	19.98	21.54
	Oreochromis variabilis	0.13	0.14
	Oreochromis niloticus	25.45	27.45
	Oreochromis leucostictus	1.17	1.27
Protopteridae	*Protopterus aethiopicus*	18.42	19.86
Clariidae	*Clarias* sp.	20.82	22.45
Anabantidae	*Ctenopoma muriei*	0.01	0.01
Cyprinidae	*Enteromius* sp.	0.03	0.03
Total		**92.74**	**100.00**

Analysis of monthly fish catch between February 2020 and February 2021 showed that March 2020 had the higher catch (13.41 mt) while October 2020 had the lower fish catch (3.49 mt) (Table 23). Although fish catch was higher in the wet season (49.82 ± 3.59 mt) than the dry season (42.92 ± 2.37 mt), the Kruskal Wallis H test showed no significant differences in inter-monthly fish catches ($H_{(11)} = 2.17$, $p = 0.998$). Similarly, one-way ANOSIM confirmed no significant seasonal differences (R-statistic = 0.04, $p = 0.259$). Permutational multivariate analysis of variance (PERMANOVA) did not yield significant seasonal differences (F = 0.50, $p = 0.642$).

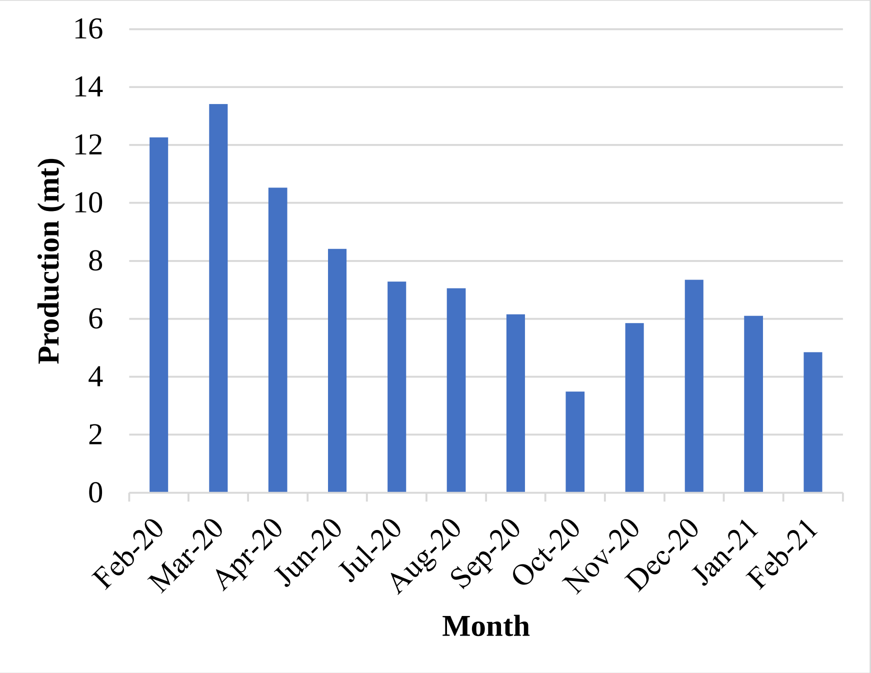

Figure 15: Monthly fish catch (mt) in Lake Kanyaboli from February 2020 to February 2021.

4.4.4. Catch per unit effort in Lake Kanyaboli from February 2020 to February 2021

The effective fishing effort ranged from 721.50 kWdays (October 2020) to 1478.52 kWdays (July and August 2020), with a mean of 1214.65 ± 230.25 kWdays (Figure 16). Likewise, catch per unit effort (CPUE) fluctuated between 4.33 kg / kWday (September 2020) and 10.68 kg / kWday (April 2020), and the mean CPUE was 6.38 ± 2.26 kg / kWday. The overall trend showed that CPUE decreased with an increase in effective fishing effort except for January and February 2021, when the effort was low, and CPUE was also low (Figure 16), attributed to bad weather.

Seasonal comparisons of mean effective fishing effort showed that the dry season had a high (1323.37 ± 200.21 kWdays) mean effective fishing effort compared to the wet season (1105.93 ± 219.51 kWdays) while CPUE was high in the wet season (7.30 ± 2.60) compared to the dry season (5.48 ± 1.58). The *t*-test did not show significant differences between the two seasons for effective fishing effort ($p = 0.178$) and CPUE ($p = 0.104$).

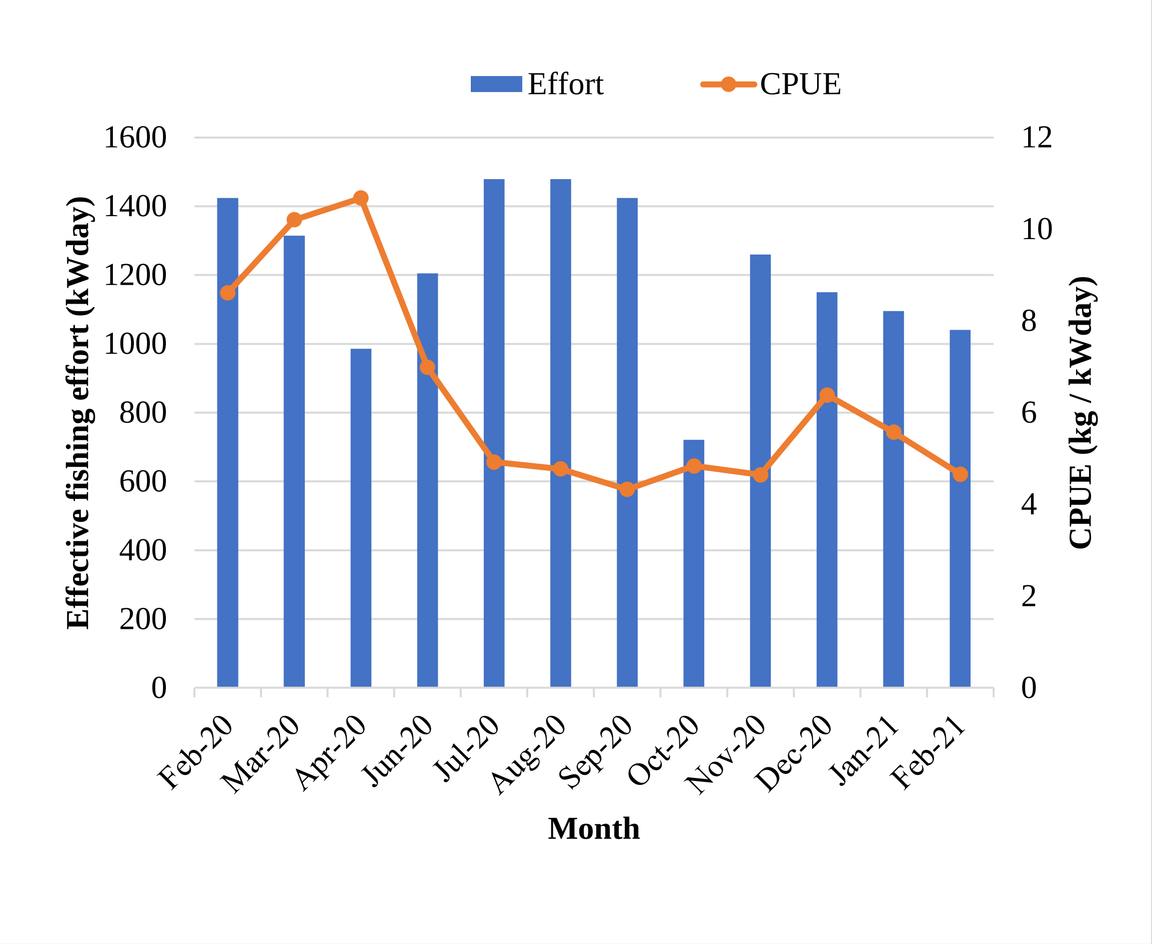

Figure 16: Monthly effective fishing effort and catch per unit effort in Lake Kanyaboli from February 2020 to February 2021. CPUE = catch per unit effort.

CHAPTER FIVE

DISCUSSION

5.1. Historical changes in the limnological characteristics and trophic status of Lake Kanyaboli

Lake Kanyaboli is a strategic aquatic ecosystem that serves many people through the provision of ecosystem services, including water for domestic use, irrigation, watering of livestock, and fisheries. The lake is also a critical biodiversity hotspot (Aloo, 2003). Despite the ecological and socio-economic importance of the lake, its ecology data is scanty. Notable studies that have previously been done on the lake include those on the fisheries and limnology (Okemwa,1981; Mavuti, 1989), the feeding ecology of *Oreochromis esculentus* (Opiyo, 1991), phylogenetic studies of haplochromines and *Clarias gariepinus* (Abila *et al*., 2004, 2008; Barasa *et al*., 2017) and a survey on its biodiversity (Aloo, 2003). The lack of detailed and consistent studies on the ecology of the lake hinders decision-making and conservation efforts.

Nevertheless, the few studies on the lake (Table 4) show that some water quality variables have been researched more (temperature, turbidity, electrical conductivity, DO, and pH) than others (chlorophyll-*a*, TSS, ammonium, phosphorus, nitrate, and nitrite). Compared to other Kenyan lakes like Lake Naivasha, Lake Baringo, and Lake Victoria, Lake Kanyaboli has received far less ecological investigations. In Lake Baringo (Ballot *et al*., 2003; Ouma & Mwamburi, 2014; Nyakeya *et al*., 2018; Walumona *et al*., 2021), Lake Naivasha (Gaudet, 1979; Gaudet & Muthuri, 1981; Kitaka *et al*., 2002), and Lake Victoria (Hecky, 1993; Kishe, 2004; Ngupula *et al*., 2012), studies of the ecological conditions are well-document. Therefore, this study gives comprehensive data on the lake's physico-chemical and Chl-*a,* as well as plankton and fish data.

Grey and published literature demonstrate that the concentrations or levels of the water quality variables are within acceptable ranges for aquatic life. While the pH is between 6 and 8, electrical conductivity has always been below 1000 μS cm^{-1} (Table 4). According to Goshu *et al*. (2017), the conductivity of most freshwaters ranges from 10 to 1000 μS cm^{-1} but can exceed 1000 μS cm^{-1}, especially in waters that are polluted or receive a lot of runoff. Their research also revealed that most naturally occurring waters had pH levels between 6.0 and 8.5. Salt lakes and groundwater brine, on the other hand, can have pH levels that are lower. Lake Kanyaboli's dissolved oxygen level has remained within the optimal range (Table 4) for fish and most aquatic life resources (Svobodová *et al*., 1993). Intermediate chlorophyll-*a* concentrations were observed in 2005 and 2020 (Table 4) (Brown & Simpson, 2001). While phosphorus was above the United States Environmental Protection Agency's (USEPA) suggested limit of below 0.1 mg/L, nitrogen species of nutrients were likewise within the non-toxic level below 10 mg/L (Fried *et al*., 2003). However, the assessment of historical trends on nutrient loading in Lake Kanyaboli was hampered by the absence of data from past studies.

Ammonium nitrogen content was found to be much higher than nitrite or nitrate in the lake according to an assessment of inorganic nitrogen levels conducted between February 2020 and February 2021 (Table 4), similar to Lake Naivasha (Muthuri & Jones, 1997). When the typical denitrification pathway is modified to one that produces ammonia under anaerobic circumstances, ammonia levels increase relative to other forms of inorganic nitrogen (Muthuri & Jones, 1997). Given that the two lakes are comparable and are both surrounded by macrophytes, Lake Kanyaboli is likely experiencing similar conditions to those in Lake Naivasha. The findings also revealed a higher nitrogen concentration in Lake Kanyaboli, possibly as a result of inflow from the catchment area and the presence of significant amounts of organic matter that were decomposing. Aquatic vascular plants produce significant amounts of organic nitrogen during decomposition (Nichols & Keeny 1973). In addition, more nutrients are derived from the excretory waste of aquatic animals and livestock that water in the lake and the Yala River inlet canal. Moreover, as the livestock grazes in the wetland, their excretion adds to the nutrients that can be carried by runoff into the lake (Iteba *et al*., 2021). Approximately 1820 herds of cattle, 497 goats, and 322 sheep are thought to be supported by Yala Wetland daily (Mwaura *et al*., n.d). The observed pattern indicates that the major form of nitrogen accessible for plant uptake in Lake Kanyaboli is ammonium ions. Therefore, it's crucial to assess the ammonium content when determining if nitrogen limits growth (Muthuri & Jones 1997).

Due to its low concentration, phosphorus is regarded as a limiting nutrient for primary production in aquatic ecosystems (Maberly *et al*., 2020). However, in tropical and subtropical areas, nitrogen, not phosphorus, primarily regulates phytoplankton growth (Havens, 2000; Maberly *et al*., 2020). For phytoplankton growth, the total nitrogen (N) to total phosphorus (P) ratio of 9:1 was used; with N:P ratios higher than 9 regarded as phosphorus-limited, whilst those with ratios lower than 9 considered as nitrogen-limited (Salas & Martino, 1991). As a result, the present study's N:P ratio (6:1) indicated that N was a limiting factor for primary production in Lake Kanyaboli. This is consistent with other studies in tropical areas that show N to be a limiting factor for primary production. (Salas & Martino, 1991; Havens, 2000; Maberly *et al*., 2020).

Lake Kanyaboli had a greater seasonal change in the physico-chemical water quality parameters than a spatial variation. This is because the lake is small, mostly well-mixed, and has short water residence times. Given the high influent, Secchi depth demonstrated seasonal changes with decreased levels (increased turbidity) during the rainy season (Srifa *et al*., 2016). The nutrient concentrations were relatively high during the dry season compared to the wet season (Figure 6). Inland waters do not often have high nutrient concentrations during the dry season, where nutrients are frequently associated with runoff from agricultural catchments (Wondie *et al*., 2007; Kilonzo *et al*., 2014; Srifa *et al*., 2016). In Lake Kanyaboli, evaporation that concentrates solutes is probably responsible for the dry season's high concentration of nutrients, while dilution would result in low quantities during the wet season. (Jones *et al*., 2009; Okech *et al*., 2018; Olele & Ekelemu, 2021). The amount of rainfall and the amounts of phosphorus and silica were found to be negatively correlated (Morabito *et al*., 2018). They attributed it to the diluted in-lake concentrations following heavy rainfall. Short water residence time in lakes like Lake Kanyaboli, which drain disproportionately vast drainage catchments compared to their surface area, can also increase the diluting effect (Cardille *et al*., 2004). Walker *et al*. (2007) noted that the characteristics of the drainage basin and the average depth of the lake or reservoir primarily regulate nutrient enrichment in these systems.

In this study, several physico-chemical factors were found to have both positive and negative associations (Table 7). Contrary to Yu *et al*. (2020), who reported that TN was negatively correlated with NO_2^-, the correlation between TN and NO_2^- as well as TN and NH_4^+ was positive. The positive association between NO_2^- and NH_4^+ may point to a nitrification mechanism that releases ammonia from cow dung and urine from animals that water in the feeder canal and Lake Kanyaboli. This is in agreement with Masese *et al*. (2020a) mesocosms study, which demonstrated that cow dung and urine combined produce increased levels of nitrified ammonia. Chlorophyll-*a* and TN revealed a relatively strong relationship with Secchi depth, as was also seen elsewhere (Bachmann *et al*., 2017). Chlorophyll-*a* and Secchi depth correlation shows that dead organic matter and algae are substantial (but not the only) particle materials in the lake that influence Secchi depth (Bachmann *et al*., 2017). The lake's total suspended (non-agal) materials are influenced by the influent canal. Similarly, the relationship between Secchi depth and TN, as well as TP, demonstrates that these nutrients limit algal growth and that algal biomass decreases light penetration.

Generally, Lake Kanyaboli showed changes in physical, chemical, and biological variables for the years 1981 to 2020. These variations are the result of human activity in the Yala Wetland (Kinaro, 2008; Owiyo *et al*., 2014), which, in the past, hindered the replenishment of the water in the lake because the influent canal was blocked, leading to water quality challenges (LBDA, 1988; Aloo, 2003). The Dominion Group of Companies' maintenance of the influent canal and an outflow in 2004 may be responsible for the concentration of water quality variables in Lake Kanyaboli that are within acceptable ranges. The influent canal and an outflow help recharge the lake water. Additionally, it's possible that the densely anchored and floating macrophytes (mostly *Cyperus papyrus* and *Phragmites australis*) assist in filtering the nutrients hence buffering the lake from excess nutrients.

5.1.1. Trophic status

The availability of nutrients that promote primary production determines the trophic state of a lake and influences its clarity and algal biomass (chlorophyll-*a* concentration). Total phosphorus levels in the current study showed hypereutrophic conditions, but chlorophyll-*a* concentrations in 2005 and 2020 demonstrated that Lake Kanyaboli is eutrophic (Table 5). Similar results from Secchi depth measurements between February 2020 and February 2021 indicated that the lake is eutrophic. However, the lake fluctuated between eutrophic and hypereutrophic conditions, according to Secchi depth data from previous studies (Okemwa, 1981; Maithya, 1998; LVEMP, 2002; Jalau, 2017). These results indicate that the lake's current eutrophic condition results from increased nutrient availability.

Additionally, the trophic status for 2020 – 2021 likely demonstrates that light attenuation had an antagonistic effect on algal development in Lake Kanyaboli. In contrast, either TN, light attenuation, or both had an impact on the growth of the algae. According to Havens (2000), the agreement between the computed values of the TSI (Chl-*a*) and TSI (SD) indicates that algae predominate in light attenuation. In addition, the lower TSI (Chl-*a*) than TSI (TP) suggests that there is less algal material present than would be anticipated based on TP

(Havens, 2000). Changes in the lake's trophic status between 1981 and 2020 observed during the current study are probably related to human activity in the catchment area (Yala Wetland).

Initially, the Yala River fed the lake after flowing through the wetland, which helped in filtering surplus nutrients before they reached the lake (Muoria *et al*., 2015), potentially reducing nutrient enrichment and suspended particles. However, the wetland's buffering effect was probably reduced by the construction of the retention dike along the lake and the influent canal from the Yala River. The situation worsened after the influent canal was destroyed, which caused little to no water flowing into the lake (LBDA, 1988; Mavuti, 1989; Aloo, 2003; Abila, 2005). Moreover, the lake lacked an outlet between the1980s and 2004 (Anyona, 1997). This might have increased atmospheric depositions and nutrients from cattle watering in the lake, which increased primary production resulting in eutrophic and/ or hypertrophic conditions. However, the influent canal and an outlet were maintained after 2004, enhancing the lake's trophic condition and water recharge. Although water quality and trophic status improved, the lake is still eutrophic, suggesting persistent nutrient input from the catchment and possibly nutrient remineralization within the lake.

5.2. Water and nutrient mass balance in Lake Kanyaboli

The water balance model for Lake Kanyaboli, Kenya, based on data from the present study, showed an imbalance between the inflow (inlet canal and rainfall) and outflow (outlet, evaporation, and evapotranspiration), with more water being lost compared to gains (Table 9). The imbalance is due to more water lost through evaporation and evapotranspiration than gains from precipitation. On the other hand, the inflow from the influent canal is slightly more than the loss in the outlet.

Nevertheless, despite the imbalance between water sources and losses, the lake showed an increased water depth and volume compared to previous studies (Anyona, 1997; Aloo, 2003; Abila, 2005). These findings indicate that groundwater inflow is a major component in the water balance of Lake Kanyaboli. It is assumed that the groundwater component is vital in the mass balance of the lake, which needs to be investigated further. Moreover, the inflow into the lake is controlled due to the water needs of the farming activities in Yala Wetland; hence water fluctuation should have been more pronounced in the absence of groundwater input. Lake Kanyaboli is fringed by papyrus macrophytes which is also another form of water loss in the lake through evapotranspiration (not measured in this study), as previously observed in studies from Lake Naivasha (Åse, 1987; Jones & Humphries, 2002) and on Northern shores of Lake Victoria in Uganda (Saunders *et al*., 2007). Therefore, the sustained water level or volume in Lake Kanyaboli, despite a combined loss of the currently measured 3 % deficit plus losses not measured (macrophyte transpiration and underground water outflow), further shows the importance of the underground water source.

In addition, the slight difference in water depth between wet and dry seasons further substantiates the assumption of significant input from groundwater inflow. These findings are consistent with Jolly *et al*. (2008) and Schwerdtfeger *et al*. (2014), who also noted that groundwater plays a vital role in the water balance of wetland lakes. This explained the relatively stable water levels in Lake Kanyaboli when it lacked inflow (Okemwa, 1981; LVB, 1988; Aloo, 2003; Abila, 2005). Similarly, Schwerdtfeger *et al*. (2014) reported that some Pantana Wetland lakes in Brazil have the quantity of groundwater inflow double that of evaporation losses hence maintaining almost the same water level despite small inflow while others dry up completely due to high evaporation as they rely on surface inflow due to negligible groundwater contribution. Furthermore, Alemu *et al*. (2020) reported water imbalances in Lake Tana, Ethiopia, due to water abstraction for irrigation and potential evaporation that exceeded annual precipitation, which could not be compensated by negligible groundwater inflow. Therefore, despite Lake Tana and Lake Kanyaboli facing evaporation challenges, the contrasting results reinforce the assumption of the importance of groundwater input into water balance in the present study.

Lake Kanyaboli showed that water inflow and outflow dynamics drive nutrient dynamics in the lake, with more nutrient fluxes introduced during the wet season compared to the dry season (Table 10). The nutrient budget indicated that the inflow contributed more to the lake's N and P nutrient loads, while more were also lost in the outflow (Figure 10). For atmospheric deposition, wet deposition contributed more nitrogen loading in the lake, while phosphorus loading was driven more by dry deposition, concurring with other regional studies (Tamatamah *et al*., 2005; Muvundja *et al*., 2009) and elsewhere (Verburg *et al*., 2018). The quantity of phosphorus transported out of the system through outflow was more than the quantity in the inflow; hence atmospheric deposition and internal loading (though not directly estimated in this study) were responsible for phosphorus loading in the lake.

Mass nutrient balances from lakes and reservoirs indicate that wet deposition is a major nitrogen source, while dry deposition is a major phosphorus source (Muvundja *et al*., 2009; Verburg *et al*., 2018). Lake Kanyaboli drains a catchment area predominantly under agricultural land use. The local communities constantly burn their fields in Yala Wetland during the dry season as part of land preparation for planting in the coming rainy season, leaving the land bare. The burning of the fields and wind erosion can be major sources of nitrogen (Lewis, 1981; Boostma *et al*., 1996) and particulate phosphorus (Artaxo *et al*., 2000; Tamatamah *et al*., 2005) to surface waters. Besides, the decomposition of organic matter can release volatile ammonia and contribute to atmospheric nitrogen deposition in the lake. The high nitrogen load in the wet deposition is because of its gaseous existence in the atmosphere, which is easily trapped by raindrops. The nutrient load into the lake derived from the catchment (through the influent canal) were higher than atmospheric deposition in the present study because of the large catchment area the lake drains.

The sedimentation or net sink was high during the wet season for nitrogen, while the lake was a strong sink of phosphorus during the dry season (Table 10). The net sedimentation for phosphorus (P) in Lake Kanyaboli was relatively lower (Figure 10) compared to other lakes, such as Lake Taihu in China (Xu *et al*., 2021). Phosphorus lacks a gaseous phase and is associated with long residence time in lakes since it is only lost via outflows or harvesting of biomass (Xu *et al*. (2021). According to Xu *et al*. (2021), 63% of P associated with external loads was retained in Lake Taihu with a water residence time of > 180 days, which minimised P losses. Retention of P from external load in Lake Kanyaboli was relatively lower possible due to shorter water residence time. This probably affects assimilation by sediments because of the lower rate of sedimentation as observed in Lake Okeechobee in the USA (Havens *et al*., 2001) hence high loads being transported out of the lake (Figure 10). Similarly, net sedimentation of nitrogen showed that Lake Kanyaboli is not a strong sink of N, perhaps due to higher denitrification (atmospheric transfer) (McCarthy *et al*. 2007) and high demand by primary producers. Alternatively, N sedimentation or net sink was underestimated in our study as N_2 was not incorporated, although it is a major input source. The warm tropical temperatures favour rapid microbial processes (Havens *et al*., 2001); hence N_2 fixation should be a major source of N in Lake Kanyaboli.

Although the present study showed that more nutrients are lost through different pathways, eutrophic conditions have persistently been observed in Lake Kanyaboli (Kondowe *et al*., 2022a). This indicates that the lake relies on atmospheric deposition and internal loading in the absence of river inflow. The persistent eutrophic conditions, despite the reduced nitrogen and phosphorus sedimentation rate, strengthen the importance of internal loading and atmospheric deposition as sources of nutrient loading for primary production. In addition, wind mixing in shallow lakes like Lake Kanyaboli encourages sediments' resuspension, leading to an input of sediment-bound N and P (Halmiton and Mitchell, 1997; Schallenberg & Burns, 2004; Verburg *et al*., 2018). Therefore, Lake Kanyaboli, a shallow lake, benefits from nutrient resuspension due to prevailing wind action (Kondowe *et al*., 2022b).

Nevertheless, the reduction of external nitrogen loading will likely cause more variability in internal N cycling due to N losses through denitrification. Thus, the dynamics in Lake Kanyaboli perhaps demonstrated the existence of internal loading from sediments and dead organic matter that help maintain biological processes. Besides, livestock watering in the lake contributes to the nutrient mass load. Verburg *et al*. (2018) noted that naturally, the reduction of external loads adversely affects the nutrient dynamics of the lake, including water quality, except in the lakes with higher internal loading and resuspension of nutrients, especially in shallow lakes. Furthermore, climate change induced extreme weather events, such as heavier rainfall and prolonged droughts, can also increase external nutrient loading into the lake. Therefore, management efforts on nutrient dynamics in the lake should also consider climate change-driven dynamics.

5.3. Plankton communities in Lake Kanyaboli

Strong seasonality was observed in the plankton (phytoplankton and zooplankton) community structure and composition in Lake Kanyaboli between February 2020 and February 2021, while minimal or homogeneous variations were observed in spatial data. Three zooplankton classes, including Branchiopoda (Cladocerans), Hexanauplia (Copepods), and Eurotatoria (Rotifers), as well as seven phytoplankton classes that included Bacillariophyceae, Chlorophyceae, Cryptophyceae, Cyanophyceae, Euglenoidae, Trebouxiophyceae, and Zygnematophyceae, were identified during the study. The phytoplankton community was dominated by Chlorophyceae, while the zooplankton community was dominated by Hexanauplia (Copepods). While zooplanktons were more diversified in the dry season than the rainy season, phytoplankton reported the same number of species in wet and dry seasons but with variable composition.

In comparison to the wet season, the dry season showed higher abundances of phytoplankton and zooplankton. The abundance of plankton (phytoplankton and zooplankton) was consistent between sites. However, the composition of phytoplankton at all sites was dominated by Chlorophyceae (Figure 11a). Rotifers dominated the inflow and outflow zones of zooplankton, whereas copepods dominated the littoral zone and open water (Figure 11b). The findings support the hypothesis that plankton (phytoplankton and zooplankton) communities in Lake Kanyaboli fluctuate more between seasons than spatially. No taxon or combination of taxa accounted for more than 80% of phytoplankton abundance. No taxon or combination (2 or 3) of taxa accounted for over 80% of phytoplankton abundance. This supports the hypothesis that the phytoplankton community in the lake is not in a state of equilibrium.

5.3.1. Dynamics in phytoplankton communities

There was a correlation between phytoplankton community composition and water quality variables in Lake Kanyaboli. *Coelosphaerium* sp., *Spirulina* sp., *Protococcus* sp., *Tetraspora* sp., *Botryococcus* sp., *Phormidium* sp., and *Crucigena* sp. were positively correlated with high nutrient concentrations and water clarity (Secchi depth). Contrarily, there was a negative correlation between nutrients and *Chilomonas* sp., *Phacus* sp., *Polycystic* sp., *Euglena* sp., *Merismopedia* sp., *Closterium* sp., *Eudorina* sp., and *Diatoma* sp. These findings possibly suggest that nutrient concentration and water clarity regulate phytoplankton communities in this lake. Cyanophyceae taxa, including *Coelosphaerium* sp., *Anabaena* sp., *Spirulina* sp., and *Phormidium* sp., are found in eutrophic, nutrient-rich waters (Loza *et al*., 2014; Sharma *et al*., 2016). However, *Merismopedia* sp. is typically found in aquatic ecosystems with high water quality, which explains the negative correlation with nutrients (Willén, 2000; Abdel-Hamid *et al*., 2014). Similar to what was found in Lake Kanyaboli, nutrient levels, conductivity, and turbidity were found to have a significant influence on the phytoplankton community structure in China's Shengjin Lake (Wang et al., 2015). Phytoplankton abundance has also been shown to positively correlate with NO_3^- and NO_2^- (Armi et al., 2010).

Significant temporal and spatial changes were observed in chlorophyll-*a*, an indirect measure of phytoplankton productivity, whereas seasonal variations were more pronounced. Chlorophyll-*a* levels were lower in open water compared to littoral zones. Phytoplankton and zooplankton numbers were also lower in open water than in the littoral zone, although the difference was not statistically significant. According to Frost & Culver (2001), nutrient availability in Lake Erie, Canada, influenced spatial phytoplankton abundance with lower numbers in open water relative to inshore areas. Thus, this further shows that littoral zones are more productive than open waters because of the frequent mixing (Wondie *et al*., 2007).

Contrary to a previous study in the same lake by Wilfred *et al*. (2005) that reported blue-green algae dominated in a sample by a relative abundance of over 50%, phytoplankton abundance in Lake Kanyaboli indicated a dominance of green algae (Table 14). In addition, many African lakes and reservoirs, like Lake Victoria and Malawi, are typically associated with blue-green algae, yet here the green algae showed dominance (El-Otify *et al*., 2003). Green algae dominance in Lake Kanyaboli indicates better water quality than in other lakes that are highly influenced by nutrient loading from agricultural and urbanization catchments (Hecky *et al*., 2010; Yongo *et al*., 2021; Niyoyitungiye *et al*., 2019). But prior studies had demonstrated that the lake had previously experienced hyper-eutrophication when the influent canal was blocked, limiting water replenishment from the Yala River except through rainfall and backflow from Lake Victoria (Abila, 2005; Kondowe *et al*., 2022a). A similar change in phytoplankton dominance from Cyanophyceae to Chlorophyceae was documented by Ballot *et al*. (2009) in Lake Naivasha because of better water quality. On the other hand, because of the declining water quality, a change from Chlorophyceae to Cyanophyceae was seen in Lake Olodien (Ballot *et al*., 2009). When nitrogen and phosphorus levels are increased, the phytoplankton community shifts from being dominated by diatoms and chlorophytes to being dominated by cyanobacteria (Hecky *et al*., 2010).

Wind turbulence could also be associated with phytoplankton composition dynamics in Lake Kanyaboli. During the study period, the lake had an average wind speed of 2.14 $m.s^{-1}$, which was slightly higher than the wind speed (2 $m.s^{-1}$) that inhibits the production of cyanobacterial (blue-green algae) blooms (Blottiere, 2015). This likely favoured the growth of green algae over cyanobacteria (Blottiere, 2015; Visser *et al*., 2015; Zhou *et al*., 2015). Because of the competitive advantage, fast-growing chlorophytes have over comparatively slow-growing cyanobacteria, the latter is more abundant owing to wind-induced mixing that resuspends nutrients (Jensen *et al*., 1994).

The phytoplankton community composition of Lake Kanyaboli (Table 14) suggested that the lake is not in an equilibrium state since no single taxon or a combination of two or three species contributed over 80% of the total abundance that existed for more than a month (Sommer *et al*., 1993). Due to a less stable diurnal mixed layer caused by greater radiative heating, which promotes dynamism in the lake processes, tropical lakes are linked to continual mixing (Lewis, 1987; Talling & Lemoalle, 1998). Furthermore, the dense floating papyrus macrophytes in Lake Kanyaboli play a role in the dynamics of the phytoplankton by consuming and using nutrients that the primary producers could have used. Other smaller and shallower water bodies have documented the effects of macrophytes on phytoplankton dynamics (Scheffer *et al*., 1993; Takamura *et al*., 2003; Fonseca & Bicudo, 2010; Fonseca & de Mattos Bicudo, 2011). Therefore, water exchange within the lake and macrophyte-driven nutrient uptake likely keep Lake Kanyaboli in a state of continual transition between turbid and clear state, avoiding the permanent dominance of selected phytoplankton species. In their study, Scheffer *et al*. (1993) found that macrophytes inhibit algal growth by reducing nutrient availability and, in some cases, by releasing allelopathic chemicals that are poisonous to algae. All of these may have contributed to a change in the composition of the plankton community, leading to a non-equilibrium condition and better water quality in Lake Kanyaboli. These results corroborate those of Hutchinson (1961), who observed that seasonal and/or annual physical, chemical, and biological, environmental changes cause non-equilibrium states in most lake systems.

5.3.2. Dynamics in zooplankton communities

Like phytoplankton, some zooplankton taxa responded to seasonal water quality changes in Lake Kanyaboli. *Asplanchina* sp., *Microcodon* sp., *Ceriodaphnia* sp., *Cyclop* sp., *Kerattela* sp., *Hexarthra* sp., and *Nauplius* sp. were correlated with high dissolved oxygen, nutrient concentrations and increased water clarity (Figure 14). Conversely, high concentrations of nutrients (TP, NH_4^+, SRP), electrical conductivity, and the water temperature had a negative correlation with *Branchionus* sp. The genus *Kerattela* sp. can be used as an indicator for eutrophic conditions, given its relationship to nutrients and Secchi depth. Rotifera species belonging to the genera *Brachionus, Polyarthra, Keratella*, and *Filinia* are bioindicators of organic pollution in inland waters (Ejsmont-Karabin, 2012; Rosińska *et al*., 2019; Zakaria & El-Naggar, 2019; Krupa *et al*., 2020). Additionally, Iskn *et al*. (2020) found that nutrients influence zooplankton directly through effects on ontogeny and physiology and indirectly through changes in food sources.

Copepods were the most abundant zooplankton in Lake Kanyaboli, followed by rotifers and cladocerans. Copepods are the most abundant in other lakes and rivers as well, including Lake Kivu (Isumbisho *et al*., 2006), Lake Victoria (Ngupula *et al*., 2010), and Lake Shinji-Ohashi and River-Lake Nakaumi in Japan (Uye *et al*., 2000). In addition to being effective feeders, their small size helps explain copepod dominance in aquatic ecosystems (Waya *et al*., 2014). On the other hand, cladocerans are more sensitive to changes in water quality; hence their low density may be caused by eutrophication or planktivorous fish predation (Zakaria & El-Naggar, 2019).

Further evidence of size-selection predation by planktivorous fishes in the lake, like the haplochromines, which prefer large-sized zooplankton, can be seen in the variability in the zooplankton community (Hansson *et al*. 2007). Consequently, more cladocerans than copepods are preyed upon (Helenius *et al*., 2015). Besides, small nauplii dominated copepods, and their size makes them unappealing to large fish; instead, they are mostly consumed by young fish (Hansson *et al*., 2007). Disparities in abundance found between zooplankton groups in the present study can be attributed to these phenomena.

5.3.3. Seasonal variation in phytoplankton and zooplankton

Seasonal variations were also observed in temporal species diversity indices for phytoplankton and zooplankton, with dry months typically recording high diversity (Table 13). This most likely demonstrates the role of seasonal changes in physico-chemical variables as a driver of the plankton community. Similarly, high dry-season plankton densities have been observed elsewhere for phytoplankton (Deng *et al*., 2007) and zooplankton (Shukla & Singh, 2013; Razak & Sharip, 2019). Seasonality in Chemical (nutrients) and physical (weather) factors and their interaction largely influence phytoplankton succession (Tian *et al*., 2013). During the dry season, phytoplankton blooms are common, which benefits the zooplankton community and biomass (Razak & Sharip, 2019). In contrast to Wang *et al*. (2015) from Shengjin Lake, China, the average phytoplankton diversity index results from the current study were lower.

Competition for nutrients between phytoplankton and macrophytes probably explains the lower diversity index in Lake Kanyaboli. Since phytoplankton and zooplankton assemblages act as water quality indicators, the low abundance of phytoplankton may also indicate a generally deteriorating ecosystem health associated with organic pollution (Pinto-Coelho *et al*., 2005; George *et al*., 2012). Taxa of *Closterium* sp., *Navicula* sp., and *Nitzschia* sp. were observed in the lake, and their presence is consistent with the introduction of organic pollution (Yusuf, 2020). Euglenoidae (*Euglena* sp. and *phacus* sp.) phytoplankton were also found in the present study, which may be an indicator of anthropogenic linked nutrient enrichment (George *et al*., 2012).

Research on other small lakes and reservoirs has shown similar dramatic seasonal changes in the abundance and composition of plankton communities, as has Lake Kanyaboli (de Souza Barroso *et al*., 2018; Kraus *et al*., 2019; Nguyen *et al*., 2019). Seasonal factors like flooding, which causes water turbidity and decreases productivity, have been linked to this phenomenon (Morabito *et al*., 2018). Additionally, the shorter residence time in smaller lakes and reservoirs is closely linked with rapid water transit (Borges *et al*., 2008; Ismail *et al*., 2019). Because of this phenomenon, phosphorus may be lost before it settles to the bottom sediments, where it can be used during resuspension, which affects nutrient availability for primary production. The combination of these factors likely mitigates the severity of algal blooms during the wet season but promotes their formation during the dry season after discharge levels decline (Srifa *et al*., 2016).

5.4. Fish and fisheries of Lake Kanyaboli

The present study investigated ichthyofauna in Lake Kanyaboli, an afro-tropical lake, to determine the fish catch, composition, abundance, catch per unit effort (CPUE) and effective fishing effort. The annual catch, CPUE, fishing effort and fish species composition between 1981 and 2020 in Lake Kanyaboli showed a declining trajectory. The one-year data (February 2020 to February 2021) showed that the wet season was associated with high fish abundance and CPUE, while the fish catch was high in the dry season. The data further revealed that the family Cichlidae dominated the catch in species diversity and abundance, while Protopteridae, Clariidae, Anabantidae, and Cyprinidae were characterised by low species diversity and abundance. The present study showed that fish abundance, catch, CPUE and community structure were related to temporal variations in fishing effort.

The fishing effort showed an increasing trend against decreasing fish catches over time (Table 18). The results further confirmed the hypotheses that fish diversity, composition and catch vary with time. These parameters (diversity, composition, and catch) are directly or indirectly influenced by human pressure and environmental variables like other tropical lakes (Tejerina-Garro *et al.*, 1998; Amarasinghe & Welcomme, 2002; Cheng *et al.*, 2010). In addition, introduced (alien) fish species such as *Oreochromis niloticus* have established themselves in the lake, while native species such as *O. esculatus* and *O. variabilis* are proportionally reducing in the catches.

5.4.1. Historical trends in fish composition, catches and effort

The historical fish catches from Lake Kanyaboli show a decreasing trend over time (Table 18). In terms of species composition, *Clarias* sp. contributed more towards total catch, but in terms of families, Cichlidae (tilapias) dominate catch composition. However, the contribution of tilapia species to total fish catch has reduced by over 30% since 2003 (Lihanda *et al*., 2003), probably due to shrinking stock size due to increasing fishing pressure and the use of illegal fishing gear (Abila & Othina, 2005). Monthly fish landings between February 2020 and February 2021 showed no statistically significant difference on a temporal scale.

Fish catches in Lake Kanyaboli, as in many other small lakes and reservoirs, have been on the decline, for instance, Lake Chapala, Mexico (Moncayo-Estrada *et al*., 2012), Kyoga Lake system in Uganda (Ogutu-Ohwayo *et al.*, 2013), and Lake Naivasha, Kenya (Yongo *et al*., 2021). This phenomenon is due to the ever-increasing fishing effort resulting from human population growth and the lack of alternative livelihoods among fishers. The number of fishers and fishing crafts in Lake Kanyaboli continues to increase against a declining CPUE (Table 18) which is probably indicative of shrinking fish biomass. The catch results are contrary to Lihanda *et al*. (2003) suggestions that the Lake Kanyaboli fishery could yield up to 432 mt annually (36 tonnes per month) because the expansion of effort in the lake has resulted in the reduction of catch. The fishing effort has probably reached unsustainable levels, hence the decline in catch and CPUE. Multispecies fishery like that of Lake Kanyaboli is associated with an increase in catch with the increase of the fishing effort until a stabilizing point when an increase in fishing effort does not lead to increased production (Lorenzen *et al*., 2006). Overexploitation (due to high fishing effort) leads to a decrease in CPUE, resulting in the reduction of catch per area and an overall decline in catches in the long run (Kantoussan *et al*., 2014). Moreover, most lakes and reservoirs are found among resource-poor communities that rely on agriculture or fisheries (Ogutu-Ohwayo *et al*., 2013).

The declining CPUE in Lake Kanyaboli is also likely linked to destructive fishing gears that indiscriminately scoop small-size *Oreochromis* spp. and *Haplochromis* spp, such as a seine net, banned by the Fisheries Department in 2003 because it was considered unsustainable and destructive fishing practice (Abila & Othina, 2005). Furthermore, OSIENALA (1998) and Abila (1998) reported the use of small, meshed gillnets of between 1- and 2-inch mesh sizes, which resulted in the exploitation of juvenile *Oreochromis* spp. (below 25 cm) potentially affecting recruitment. Furthermore, recruitment overfishing is also driven by selectively targeting large-bodied fish such as *Oreochromis niloticus* due to the market preference for large-sized fish. Abobi *et al*. (2019) noted that the Tono reservoir (Ghana) fishery was driven by market preferences of large-sized *O. niloticus*, primarily spawners, potentially resulting in recruitment overfishing of the reservoir. Lake Kanyaboli's fishers use of smaller mesh size gillnets ($^1/_2$") either for haplochromines or as an adaptation to increase catch also poses a risk of growth overfishing on big-sized fish species such as *O. niloticus* and *O. esculentus* concurring with findings of Yongo *et al*. (2021) in Lake Victoria and Ahmed *et al*. (2001) in India's Kaptai Reservoir.

Besides the decline of catch and CPUE, fishing activities in Lake Kanyaboli, such as open access, threaten fish diversity. Despite the lake being managed under co-management, the existing Beach Management Unit (BMU) does not enforce existing regulations. Registration as a fisher is the only condition to participate in the fishery. The reliance on BMU to perform all management activities weakens the management system since such committees are dominated by fishers who are interested parties. Similar weaknesses and laxity, coupled with corruption of fisheries officers, have been reported in Lake Victoria, leading to the overexploitation of some species (Njiru *et al*., 2008). Operating a fishery under open-access arrangements is disadvantageous to the maintenance of fish diversity, the overall sustainability of a fishery, and the local economy's well-being since it leads to the collapse of fish stocks and dissipation of economic rent (Kasulo, & Perrings, 2006; Arthur, 2020).

5.4.2. Fish species diversity, abundance catch and catch per unit effort changes between February 2020 and February 2021

Haplochromines were more diverse, followed by tilapia species in catch composition disintegrated data. Fish diversity, abundance, catch, and CPUE did not vary significantly between the dry and wet seasons during the February 2020 and February 2021 study period, while fishing effort was significantly different between the wet and dry seasons. Although fish diversity, abundance, catch, and CPUE did not show seasonality, fish abundance and catch were high in the wet season, while CPUE was high in the dry season.

Fish biomass (a function of catch) peaks during flood months (wet season) due to increased productivity and availability of more habitat (Madsen & Shine, 2000; Soyinka *et al*., 2010; Jin *et al*., 2019). Rainfall is associated with the loading of nutrients into the lake, resulting in high primary production at the base of the food web, favouring higher trophic levels (Mulimbwa *et al*., 2014). Therefore, the abundance of fish in this study during the wet season is linked to the increased food supply during the onset of rains and the late wet season associated with the plankton boom, while the overall plankton abundance favoured the dry season (Kondowe *et al*., 2022b). In addition, the wet season increases the wetted area, which expands the littoral zone rich in macrophytes that provide refuge to fish from predators (Warfe & Barmuta, 2006) and act as nursery and feeding areas (Sabo & Kelso, 1991; Bulla *et al*., 2011; Ryken *et al*., 2015) making them susceptible to fishing which is also operated in the shallow waters.

The high catch in the wet season was probably because of turbid conditions (due to suspended particles) from the influent canal that influence fish activity and sight resulting in a high probability of fish being caught by fishing gears. The results corroborate with studies by Prchalová *et al*. (2010) and Mensah *et al*. (2018), who reported high fish catches in the wet season attributed to turbidity that influences fish activity and ability to detect fishing gear. Turbidity causes an earlier decrease and later increase of light intensity in turbid water at sunset and sunrise, affecting the diel activity of fish (Prchalová *et al*., 2010). Moreover, high fish activity is observed in a turbid environment (Andersen *et al*., 2008; Hazelton & Grossman, 2009) hence susceptible to capture by gillnet and other stationery gear (Prchalová *et al*., 2010). However, contrary results have been reported by Halls & Welcomme (2004) in the Mekong floodplain and Ayanwale *et al*. (2013) in Lake Tagwai, Minna, Nigeria, where catches were high in the dry season due to reduced water levels which reduces macrophyte refuge and limits dispersal of the fish into wider area exposing them to capture. Likewise, Abila & Othina (2005) also noted that fishermen tend to get higher catches of *Protopterus* spp. and *Clarias* spp. during the drier months, the lake water recedes in Lake Kanyaboli, exposing fish that hide in papyrus vegetation. However, the lake showed less variability in water depth during the present study because of influent canal maintenance (that was nonfunctional before 2005), which is an additional water source besides groundwater inflow that replenishes the lake.

The results in the present study (from February 2020 to February 2021) also showed that CPUE was higher in the wet season than in the dry season. The high CPUE is likely due to reduced fishing effort during the wet season because of bad weather, such as windy and rainy conditions. Moreover, the fishing effort in terms of the number of days spent fishing reduced while fish catches peaked during the wet season. Shirakihara *et al*. (1992) noted that CPUE increases amidst reducing fishing effort when fish abundance remains unchanged in the short-term. The high CPUE in the wet season could also be due to turbid conditions, as corroborated by de-Graaf & Ofori-Danson (1997), who noted that CPUE increases with turbidity attributable to the inability of the fish to detect fishing gears resulting in high catch with the same fishing effort.

The present study further revealed that species considered virtually extinct in Lake Victoria, especially *Oreochromis esculentus* occur abundantly in Lake Kanyaboli while *O. variabilis* numbers are dwindling. Yet, studies on fish abundance, diversity and distribution, as well as the catch and effort dynamics, have mainly focused on Lake Victoria (Okaronon, 1994; Bundy & Pitcher, 1995; Kitchell *et al*., 1997; Naigaga *et al*., 2011; Witte *et al*., 2013; Outa & Yongo, 2017). However, anthropogenic activities such as the conversion of Yala Wetland and agricultural expansion and intensification in the catchment area of Lake Kanyboli pose threats to the lake as potential sources of sediments and nutrients. Nutrient input can result in eutrophication and the proliferation of toxic cyanobacteria (Yongo *et al*., 2021). Sitoki *et al*. (2012) indicated that the replacement of *Aulacoseira* and diatoms by blue-green algae caused the decline in *O. esculentus* and *O. variabilis* in Lake Victoria.

5.4.3. Management implications of the fish composition and production study

The fish species composition of Lake Kanyaboli before the reclamation of part of Yala Wetland was not documented. This situation informed concerns about the potential for the disappearance of fish species unnoticed (Aloo, 2003). The few available studies on species composition in the lake post-Yala Wetland reclamation show temporal variations (Okemwa, 1981; Maithya, 1998; Aloo, 2003; Masai *et al*., 2005). When comparing the present and the previous studies, it is evident that *Coptodon zillii,* which formed part of the catch in 1981(Okemwa, 1981), is non-existent. Furthermore, *Haplochromis martini, Oreochromis variabilis, Labeobarbus* sp., *Xenoclarias* sp., and *Ctenopoma muriei* are hardly encountered in the catch and are becoming scarce. An isolated individual of *Bagrus* sp. (only one throughout the study) was discovered during the 2020 survey, but the species has not been previously reported to inhabit the lake, possibly indicating new introductions from the catchment facilitated by a direct connection between the Yala river and the lake with the aid of the influent canal constructed during reclamation of part of Yala Wetland. The *Bagrus* sp. are distributed among other areas in Lake Victoria and its catchment basin (Seegers *et al*., 2003; Masese *et al*., 2020b), possibly finding its way to Lake Kanyaboli through the Yala river channel.

Introducing new fish species is problematic if they compete for the same niche with native species and the non-native species are ecologically more versatile. The trend is worrying since fish species invasions may have disastrous consequences if unchecked. For example, it is estimated that nearly 200 haplochromines and one tilapiine species (*Oreochromis esculentus*) have been lost in Lake Victoria due to the introduction of predatory Nile perch (*Lates niloticus*) (Kudhongania *et* al., 1992; Hickley *et* al., 2008; Outa *et* al., 2020). Therefore, it is important to safeguard Lake Kanyaboli, which has substantial biomass of *O. esculentus* and six haplochromine species (*Lipochromis maxillaris, H. (Astatotilapia)* *nubilus, Astatotilapia* sp. *"bigeye"*, *Pseudocranilabrus multicolour victoriae*, *Xystichromis phytophagus* and *Astatoreochromis allua*iu*di*) that once formed part of the catch in that lake and currently showing signs of resurgence (Njiru *et al*., 2010). Nevertheless, the number of haplochromine species has remained constant over the years. Colonisation by alien fish species has already affected most of the Lake Victoria basin, such as Nile Perch (*Lates niloticus*) in one of Yala Wetland's Lake Sare (Gichuki *et al*., 2005), and several cyprinids have established themselves in rivers (Masese *et al*., 2020b) hence posing more threat on Lake Kanyaboli.

Furthermore, some fish species (*Oreochromis esculentus* and *O. variabilis*) are "Critically Endangered" (Table 19) (IUCN, 2019). However, *O. esculentus* occur in substantial quantity compared to *O. variabilis,* possibly indicating replication of competition from *O. niloticus* that they (*O. esculentus* and *O. variabilis*) faced in Lake Victoria (Njiru *et al*., 2010). Likewise, *O. niloticus*, which has established itself in Lake Kanyaboli and most of the Yala Wetland system, threatens the existence of native tilapiine species due to potential hybridisation with *O. esculentus* and *O. leucostictus* (Nyingi & Agnèse 2007; Angienda *et al*., 2011; Ndiwa *et al*., 2014). On the other hand, *Schilbe* sp. and *Ctenopoma muriei* are scarce in the catches and occur in limited biomass in Lake Kanyaboli (Table 19) despite being common in other Lake Victoria Basin systems (Masese *et al*., 2020b). The low yields of these species perhaps entail overexploitation and adverse effects of human activities in the drainage basin of Lake Kanyaboli. Activities such as intensive farming contribute to eutrophication due to the extensive use of fertilizers, risking the lake's fish biodiversity.

Furthermore, the lake faces more potential pressure from developers expanding tourism, with one resort already operational and two more under construction projects posing a further likelihood of interference if not properly regulated. Previous studies have documented the effects of anthropogenic activities in various lake drainage basins on the existence of fish biodiversity both in Kenya (Aloo, 2003; Kiage *et al*., 2009; Yongo *et al*., 2021), regional (Jamu *et al*., 2011; Amoutchi *et al*., 2021) and elsewhere (Moiseenko *et al*., 2009; Debjit *et al*., 2010; Sharip & Jusoh, 2010; Wang *et al*., 2015). Therefore, fish biodiversity threats faced by the lake are generic and global due to trade-offs between conserving the environment and the social and economic needs of the countries. Besides, there are signs of overfishing due to high fishing pressure in the lake to satisfy the high demand for fish and the absence of alternative economic activities among riparian communities. In Africa, overfishing is a recurrent problem closely tied to environmental conditions. When conditions are not conducive for traditional agriculture, most people around water bodies resort to fishing increasing fishing pressure (Balirwa *et al*., 2003).

CHAPTER SIX

CONCLUSIONS AND RECOMMENDATIONS

6. Conclusions

The present study has comprehensively looked at the state of Lake Kanyaboli and summarized key findings to help bridge the lack of water quality data on the lake. The study forms a basis for future studies to ensure progressive data-driven decision-making. Overall, Lake Kanyaboli is under increasing pressure from human activities in the catchment area and on the shores of the lake that affect its water quality and quantity, nutrient dynamics, and plankton and fish communities. Below are conclusions based on the objectives of the study.

6.1.1. Historical changes in the Limnological characteristics of Lake Kanyaboli

Lake Kanyaboli's physical, chemical, and biological features have all been significantly impacted by human activities over time. Limnological characteristics and trophic status showed temporal variations, but some variables are more researched (temperature, SD, EC, DO, and pH) than others (chl-*a*, SRP, NH_4^+, NO_3^- and NO_2^-). The lake also showed eutrophic conditions based on Secchi depth, total phosphorus and chlorophyll-*a*. Because of rising human and livestock populations in the lake's catchment and surrounding areas, eutrophication of the lake is becoming an increasingly serious problem. The catchment is used to grow crops with varying nitrogen, phosphorus, and potassium fertilizer requirements. These nutrients could worsen the lake's already eutrophic condition if left unchecked. The lake has significant socio-economic importance to local communities. Therefore, it is crucial to maintain regular monitoring of water quality and the lake's ecological status to ensure the long-term viability of ecosystem goods and services derived by the local populace. The lake should be managed and conserved to ensure its continued function as a refugium for endangered fish species, some of which are remnant cichlids from Lake Victoria's original population.

6.1.2. Nutrient mass balance in Lake Kanyaboli

The current inflow and outflow regime of nutrients indicate that Lake Kanyaboli receives more nutrient loads than it losses through the outflow channel. Furthermore, in the absence of inputs from the Yala River through the influent canal, which balances losses through the outflow canal, the internal nutrient loading and atmospheric deposition can sustain primary production in the lake. Therefore, in the case of eutrophication, reducing the inflow nutrient load alone is not enough to curb the problem in the absence of outflows. Although I never collected relevant data, the lake sediments are potentially a major sink and sequestrate N and P that play important roles in the remineralization. Management of nutrient loading should also target holistic approaches that address catchment management in the Yala River and Lake Kanyaboli drainage basin and land use management.

6.1.3. Phytoplankton and zooplankton communities in Lake Kanyaboli

The results of this study demonstrated that physico-chemical variables influence the diversity and composition of phytoplankton and zooplankton in a small tropical lake. The lake's physico-chemistry and ecology are influenced by seasonality, as evidenced by variations in water clarity (Secchi depth), dissolved oxygen concentration, and nutrient concentrations (ammonium, nitrate, nitrite, and total nitrogen), which were higher during the dry season. Phytoplankton communities in the lake demonstrated that the system is not in an equilibrium state and that plankton community in the lake fluctuate with the seasons. Indicators of nutrient enrichment were found to be the phytoplankton taxa *Navicula* sp. and *Anabaena* sp., while taxa like *Merismopedia* sp. could be used as bioindicators of high water quality. Two species of zooplankton, *Kerattela* sp. and *Cyclops* sp., are useful indicators of eutrophication because they thrive in areas with high nutrient concentrations. Food web components like zooplankton and fishes, as well as the lake's complex heterogeneous physical and chemical variables, were crucial in keeping the lake in a non-equilibrium state through the regulation of algal blooms of selected species. This study shows that long-term monitoring of small tropical lakes' water quality and biological communities is crucial to deciphering their structural and functional dynamics. This study helps understand the drivers of biological community structure and composition in small, human-influenced tropical water bodies.

6.1.4. Fishes and fisheries in Lake Kanyaboli

The findings of this study have indicated the importance of long-term evaluation of fish species composition, catch and CPUE of small tropical lakes and reservoirs. Fish species diversity, composition, catch and CPUE varied temporally in Lake Kanyaboli as a function of seasonality and fishing effort. The results from February 2020 to February 2021 survey also showed long-term inter-monthly variation in the catch, species composition, CPUE and fishing effort, which has been lacking. Moreover, the study has demonstrated the ecological significance of Lake Kanyaboli in conserving threatened fish species that once formed part of Lake Victoria's original fish community. However, the lake's fish are threatened by illegal fishing gears and methods, anthropogenic activities in the catchment area, and potential introductions (*O. niloticus* and *O. leucosticus*) that may outcompete native species.

Similarly, the lake's fish stocks are likely experiencing overexploitation, as suggested by declining catches and CPUE, and the disappearance of some fishes such as *Coptodon zillii*, *Enteromius* sp., *O. variabilis*, *schilbe* sp. and *Ctenopoma muriei* due to increasing fishing effort. Perhaps multiple stressors are acting on Lake Kanyaboli, but a lack of strong management structures, such as enforcing fisheries regulations, exacerbates the problem. Therefore, there is a need for improved enforcement to curb illegal fishing gear and regulate the number of fishers to ease fishing pressure. Furthermore, continued monitoring of fish diversity and population dynamics in the lake is vital to ensure timely remedies.

6.2. Recommendations

This study proposes the following recommendations that should be given priority, among other measures, to safeguard the sustainability of Lake Kanyaboli and its catchment area:

i. Lake Kanyaboli needs constant water quality monitoring to assess the suitability of lake water for biological communities and enrich the lake's literature. More research is required because there are discrepancies and gaps in the data collected over time regarding nutrient concentrations as well as biological factors like plankton composition and chlorophyll-*a* concentration. Strengthening the Yala Wetland National Reserve status is vital to ensure sustainable use of the wetland and Lake Kanyaboli.

ii. Nutrient management in Lake Kanyaboli should focus on input and output dynamics. For instance, in case of nutrient accumulation in the lake, an increase in discharge would help flush the system due to a reduction of water residence time. Similarly, there is a need to ensure constant inflow and outflow in the lake to sustain the flushing effect through water exchange.

iii. There is a need to constantly monitor plankton abundance dynamics to understand the long-term effect of the changing environmental conditions and water recharge in the lake on plankton communities and the long-term equilibrium state of the phytoplankton. Moreover, plankton communities are driven by seasonality in environmental conditions; hence, mechanisms that influence environmental conditions, such as influent, are vital in maintaining a non-equilibrium state.

iv. Fisheries management in Lake Kanyaboli needs strengthening to strike a balance between fishing effort and catches because catches and CPUE showed a declining trend between 2003 and 2020 while fishing effort increased during the same period. Open-access fishing and the use of illegal fishing gear persist despite the presence of participatory fisheries management structures (beach management units) all around the lake. Therefore, timely intervention is needed to manage the fishing effort.

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APPENDICES

Appendix I: Nutrient concentration calculation equations

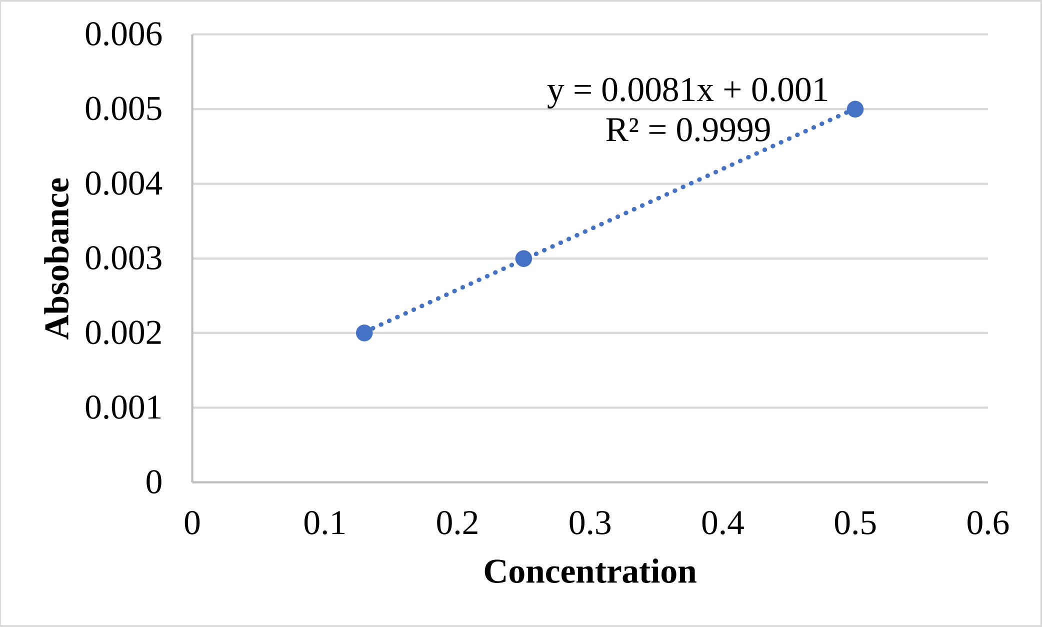


Figure I(a): Ammonium concentration equation

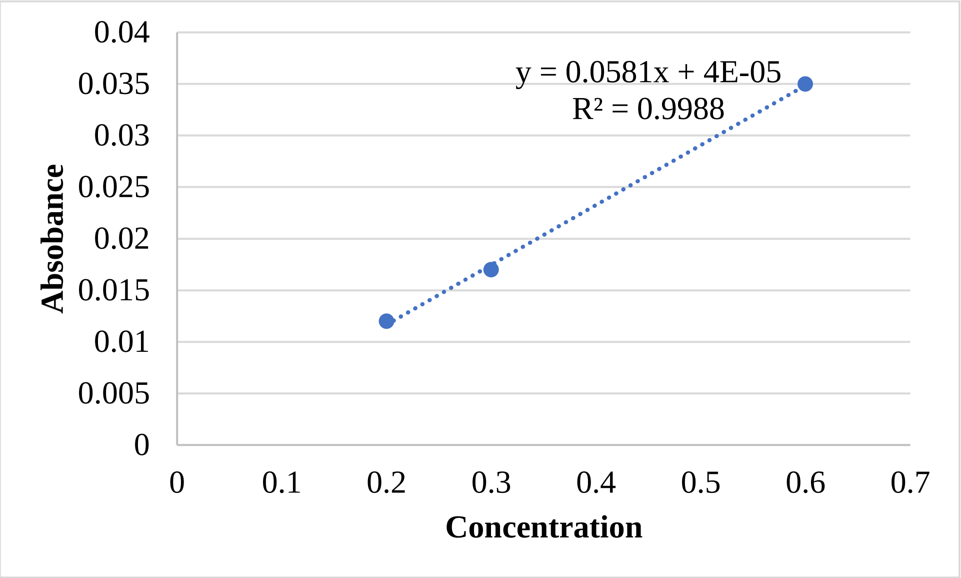


Figure I(b): Nitrate concentration equation

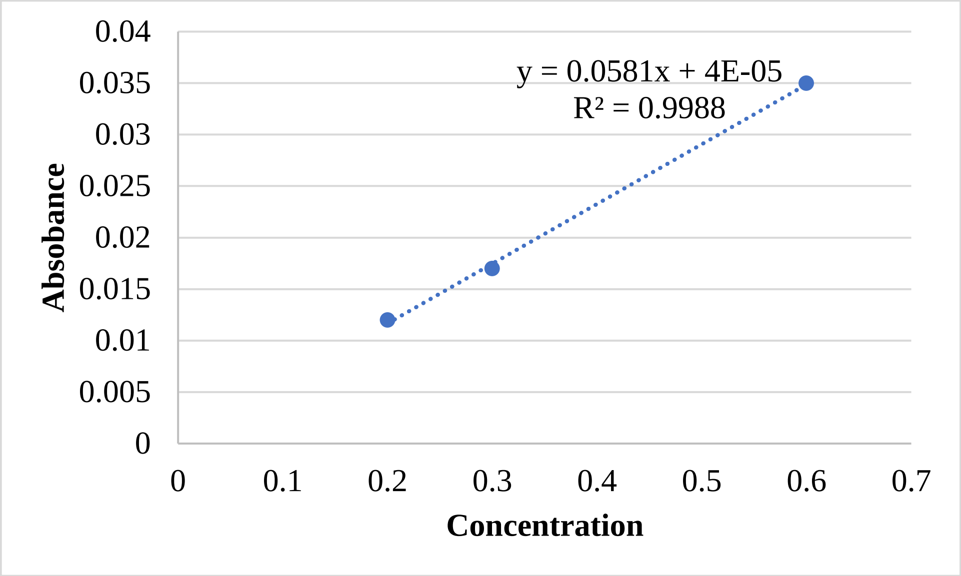


Figure I(c): Nitrite concentration equation

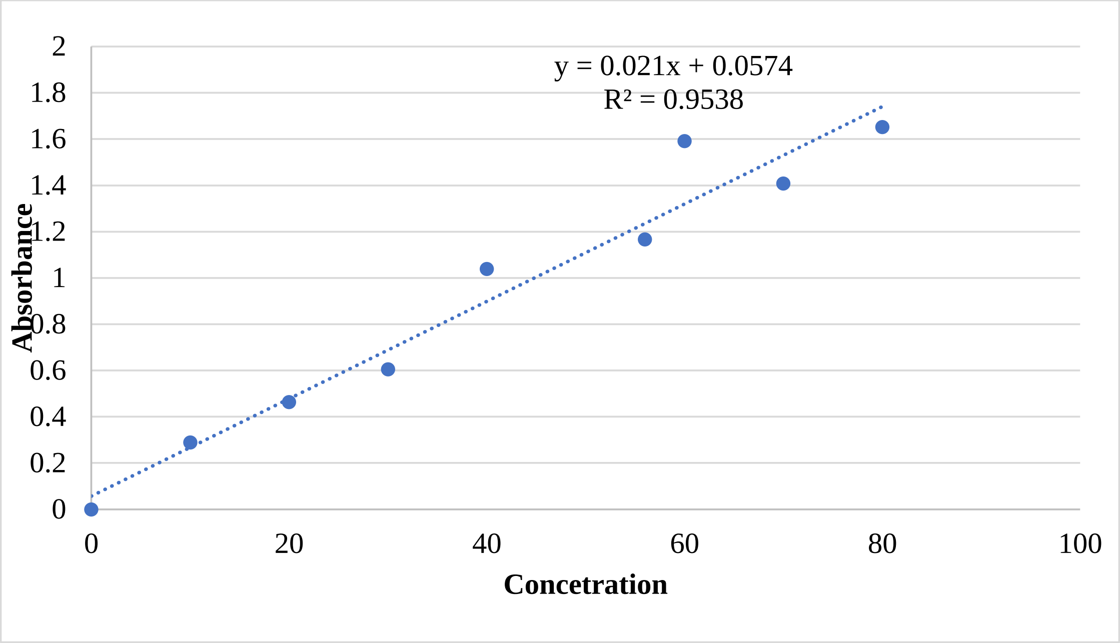


Figure I(d): Total nitrogen concentration equation

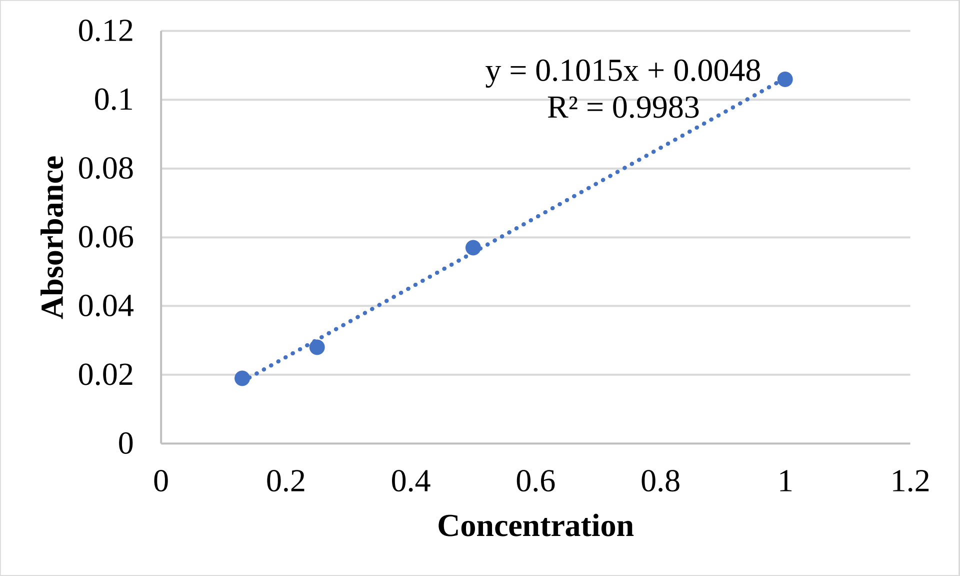


Figure I(e): Soluble reactive phosphorus concentration equation

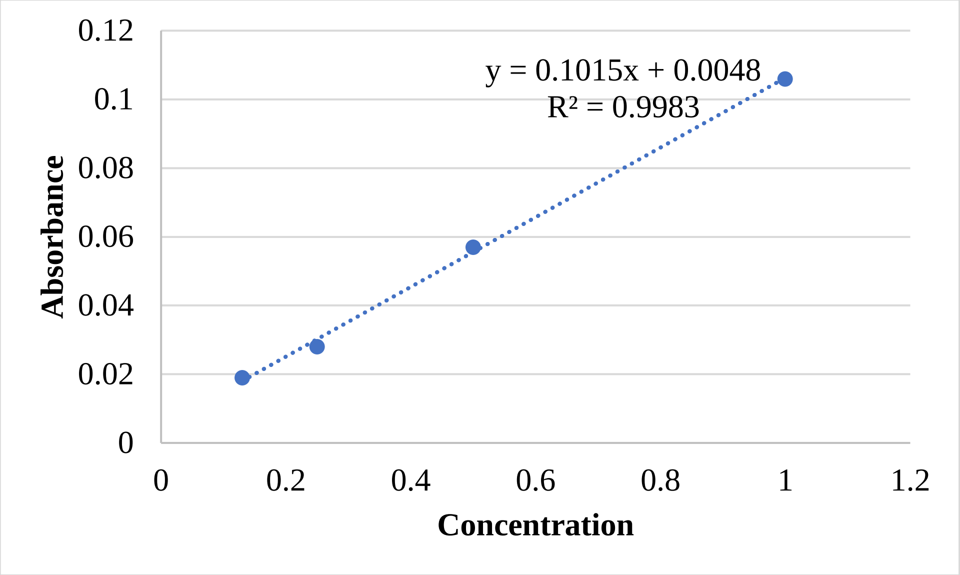


Figure I(e): Total phosphorus concentration equation

Appendix II: Levene homogeneity test for 12 selected variables measured in Lake Kanyaboli, Kenya, during February 2020 to February 2021 sampling period

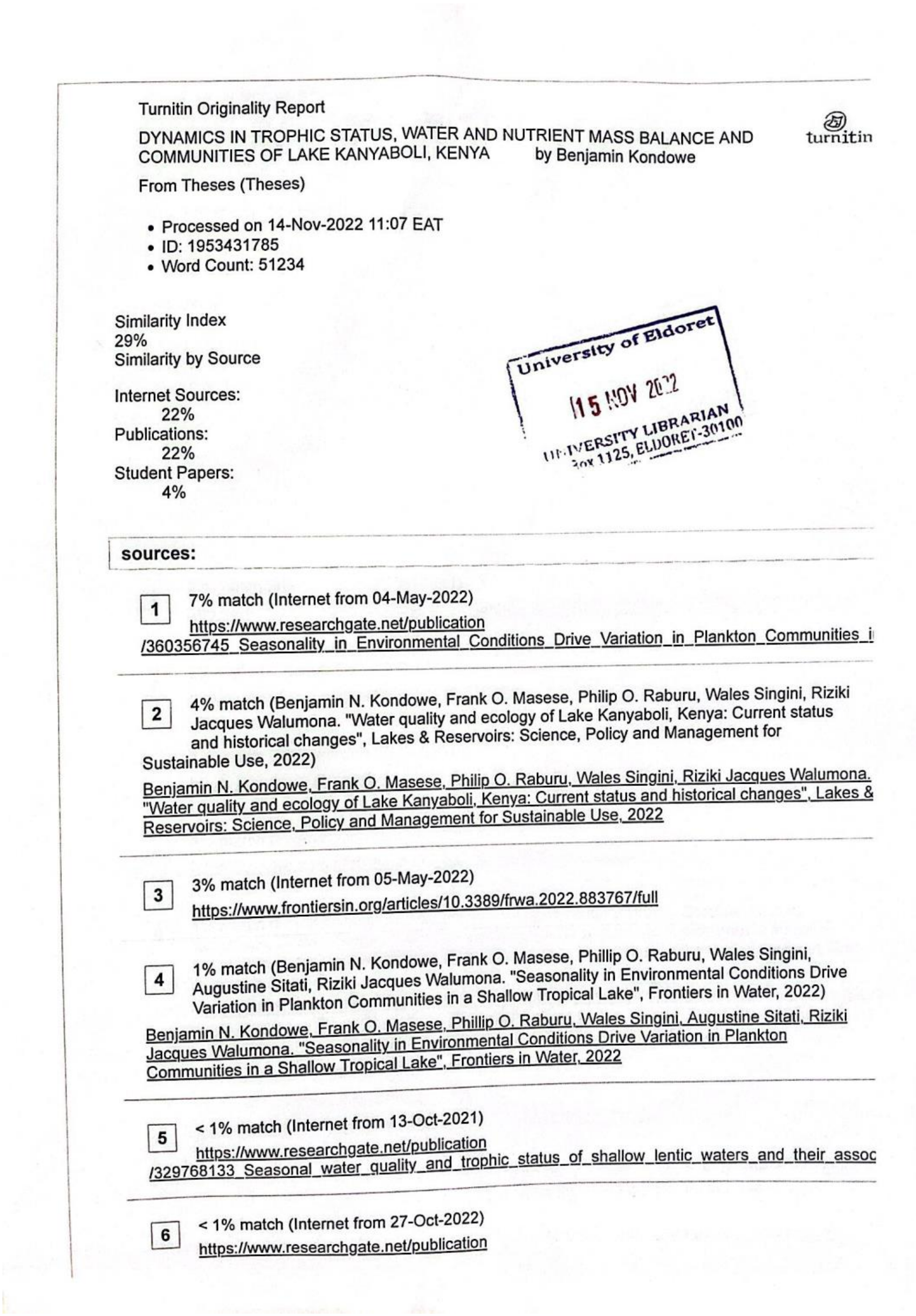
Variable	*DF*	*F*	*p-value*
SD	5	0.13	0.984
Temperature	5	0.85	0.520
DO	5	0.46	0.806
EC	5	0.73	0.602
pH	5	0.92	0.476
NH_4^+	5	0.09	0.994
SRP	5	0.10	0.991
NO_3^-	5	0.32	0.900
NO_2^-	5	0.20	0.960
TN	5	0.50	0.775
TP	5	2.29	0.057
Chl-*a*	5	1.66	0.174

Appendix III: Two-way ANOVA results for 12 variables measured in Lake Kanyaboli, Kenya, during February 2020 to February 2021 sampling period

Variable	*Source of variations*	*SS*	*DF*	*MS*	*F*	*p-value*
a) SD	Stations	0.11	5	0.02	0.60	0.698
	Seasons	0.45	1	0.45	12.69	**0.008**
	Interaction	0.06	5	0.01	0.32	0.901
	Within	1.91	54	0.04		
	Total	2.52	65			
b) Temperature	Stations	0.002	5	0.0004	0.88	0.499
	Seasons	0.0003	1	0.0004	0.94	0.335
	Interaction	0.001	5	0.0002	0.60	0.702
	Within	0.02	54	0.0004		
	Total	0.03	65			
c) DO	Stations	0.02	5	0.004	0.46	0.806
	Seasons	0.05	1	0.05	4.71	**0.034**
	Interaction	0.01	5	0.001	0.15	0.978
	Within	0.54	54	0.01		
	Total	0.62	65			
d) EC	Stations	0.0001	5	0.00003	0.17	0.971
	Seasons	0.00003	1	0.00003	0.22	0.644
	Interaction	0.0001	5	0.00002	0.15	0.978
	Within	0.01	54	0.0001		
	Total	0.01	65			
e) pH	Stations	0.01	5	0.002	0.54	0.743
	Seasons	0.003	1	0.004	0.92	0.343
	Interaction	0.01	5	0.002	0.55	0.740
	Within	0.23	54	0.004		
	Total	0.26	65			
f) NH_4^+	Stations	2.86	5	0.57	0.91	0.483
	Seasons	3.32	1	3.32	5.28	**0.026**
	Interaction	1.07	5	0.21	0.34	0.887
	Within	33.97	54	0.63		
	Total	41.95	65			
g) SRP	Stations	0.90	5	0.18	0.46	0.805
	Seasons	1.14	1	1.14	2.91	0.094
	Interaction	0.65	5	0.13	0.33	0.892

	Within	21.10	54	0.39		
	Total	24.26	65			
h) NO_3^-	Stations	0.07	5	0.01	0.10	0.993
	Seasons	0.51	1	0.51	3.69	0.060
	Interaction	0.24	5	0.05	0.35	0.878
	Within	7.39	54	0.14		
	Total	8.21	65			
i) NO_2^-	Stations	0.38	5	0.08	0.51	0.764
	Seasons	0.97	1	0.97	6.61	**0.013**
	Interaction	0.12	5	0.02	0.16	0.976
	Within	7.96	54	0.15		
	Total	9.37	65			
j) TN	Stations	0.21	5	0.04	1.93	0.104
	Seasons	0.96	1	0.96	44.12	< **0.001**
	Interaction	0.10	5	0.02	0.94	0.466
	Within	1.18	54	0.02		
	Total	2.39	65			
k) TP	Stations	0.09	5	0.02	0.40	0.847
	Seasons	0.01	1	0.01	0.13	0.715
	Interaction	0.04	5	0.01	0.15	0.978
	Within	2.51	54	0.05		
	Total	2.67	65			
l) Chl-*a*	Stations	0.78	5	0.16	84.26	**< 0.001**
	Seasons	0.02	1	0.02	12.28	**0.002**
	Interaction	0.01	5	0.002	0.96	0.462
	Within	0.04	24	0.002		
	Total	0.86	35			

Appendix IV: Similarity report



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