MODELLING LAKE LEVEL VARIATIONS, WATER BALANCE, AND FISHERIES OF LAKE BARINGO, KENYA

BY

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FEBRUARY, 2022

DECLARATION

Declaration by the Candidate

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DEDICATION

To my Almighty God who ever does miracles,

To my beloved wife, ***Hukasi Musibira Micheline***, and **our children**.
This is for the many sad and joyful moments in your individual and collective lives, that I was physically absent during the period of this study.

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To the memory of my late Father, ***Symphorien M. Mwapu***, and his beloved wife, my mother, ***Espérance M. Katundji***, and all my siblings, for their "great love".

This is my only way to acknowledge and appreciate all your efforts and sacrifices from primary education to this educational level.

Your encouragement got me this far.

The fear of the Lord is the beginning of knowledge (Proverbs 1:7)

ABSTRACT

Lake Baringo has the worst eroded catchment area and whose water levels (WLs) have been fluctuating at inter- and intra-annual time scales. These changes have been related to natural events and to human activities in the lake catchment. Changes in Lake Baringo water levels and water quality have affected fisheries production, ecological functioning and livelihoods. However, like for most tropical lakes, little is known about the linkages between WL changes, water quality parameters, fisheries production, and water/nutrient balance components of Lake Baringo. This study aimed to bridge this gap by modelling the linkages between WL variations, fisheries production, water quality, and water balance parameters of the lake. The study used both short and long-term datasets. Short-term data were generated through monthly sampling of nine stations in the lake that extended from January 2020 to June 2021 while, long-term data (1956-2018) were sourced from grey and published data on the lake. Water samples were analyzed for selected physico-chemical variables using the standard methods for the analysis of water and wastewater at the Kenya Marine and Fisheries Research Institute (KMFRI) Baringo and Kisumu laboratories. Water balance of the lake was modeled through determination of input (river discharges, rainfall) and output (evaporation, abstraction, seepage) components of the balance. Lake level changes were analyzed based on annual and monthly deviations from long-term average (DLTM) and from patterns of lake level amplitude obtained from the annual/monthly maximum and minimum lake levels (WLamp). The fisheries and ecological functioning of Lake Baringo were described using the Ecopath mass-balanced model. Four main inputs required in the Ecopath model were used and included: biomass (B) production/biomass ratio (P/B), consumption/biomass ratio (Q/B), and ecotrophic efficiency (EE). Three annual Ecopath models (1999, 2010, 2020) were generated in order to compare the temporal trend in ecosystem functioning of the lake. Statistical approaches used to analyze the data majorly included; analysis of variance (ANOVA) applications, Locally-Weighted Scatter Plot Smoother (LOWESS), linear and waveform regression analyses, and Principal Component Analysis (PCA). Results indicated that, at inter-annual scale, the lake's trophic status shifted from eutrophic to mesotrophic status at long-term (2008-2020) and short-term (2020-2021) scales, following the Carlson's Trophic Status Index (CTSI) values. Water balance modelling indicated the water inputs from the runoff contributed for 75% in 1970-1995 and 71% in 2008-2021 to the lake's storage while, direct precipitation into the lake contributed 25% in 1970-1995 and 29% in 2008-2021 to the lake's storage, indicating that runoff is the major input component of Lake Baringo's water balance. The water losses from the lake were contributed by evaporation (59-42%), abstraction (29-7%) and underground seepage (7-3%) during 1970-1995 and 2008-2021, respectively. Waveform regression significantly modeled DLTM and showed a 20-year oscillation between peak water levels in the lake. There were significant positive correlations between Water Level Fluctuations (WLFs) and both the water quality variables and Water Quality Index (WQI), and between fishery yields and WLFs in Lake Baringo. Three annual Ecopath models (1999, 2010, 2020) confirmed three trophic levels for the lake and suggested a strong bottom-up control in the lake's food web. *Oreochromis niloticus baringoensis is* modeled as the keystone species in the lake. The Ecopath network analysis for the three models provided the ecosystem functioning and fisheries indicators whose temporal trends are variable and are described in the thesis. Overall, it is recommended that water resource management policies should guide the uses of water from the lake based on WHO guidelines. Water quality assessment, WLFs, species keystonness and other Ecopath results should be considered in the application of holistic and integrated lake basin management (ILBM) approaches in Lake Baringo and its watershed in order to sustain ecological services and the lake-dependent livelihoods.

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LIST OF ABBREVIATIONS, ACRONYMS, AND SYMBOLS

a.m.s.l.:	Above mean sea level
ANOVA:	Analysis of variance
BOD:	Biological Oxygen demand
COTRA:	Collaborative Training in Fisheries and Aquaculture in Eastern, Central, and Southern Africa
CTSI:	Carlson Trophic State Index
DLTM:	Difference from the long-term mean
DOC:	Dissolved Organic Carbon
EARVL:	East African Rift Valley Lakes
EE:	Ecotrophic Efficiency
EU:	European Union
EwE:	Ecopath with Ecosim
GE:	Gross Efficiency
ILBM:	Integrated Lake Basin Management
OPI:	Organic Pollution Index
KMFRI:	Kenya Marine and Fisheries Research Institute
NPP:	Net Primary Production
NTU:	Nephelometric Turbidity Unit
OECD:	Organization for Economic Cooperation and Development
OI:	Omnivory Index
PAR:	Photosynthetic Available Radiation
PCA:	Principal Components Analysis

PET:	Potential Evapotranspiration
SD:	Secchi Disk
TDS:	Total Dissolved Solids
TL:	Trophic Level
TN:	Total nitrogen
TP:	Total phosphorus
TSI:	Trophic State Index
TSS:	Total suspended solids
TST:	Total system throughput
UoE:	University of Eldoret
WARA:	Kenyan Water Resource Authority
WHO:	World Health Organization
WLF:	Water Level Fluctuations
WQI:	Water Quality Index

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CHAPTER ONE

INTRODUCTION

1.1 Background of the study

Lakes and rivers offer a variety of ecological services and support livelihoods. Despite their importance, freshwater ecosystems are threatened by many factors including, invasive species, climate change, and human impacts (Dudgeon *et al.*, 2006). Closed lakes such as Lake Baringo (Kenya), are particularly vulnerable ecosystems because their hydrological balance is mostly controlled by evaporation due to the absence of a sufficient drainage outflow (Dunkley *et al*., 1993; Verschuren, 2003; Okech *et al.* 2019). Thus, closed lakes are very sensitive to changes in air temperature and precipitation, and in this way, they deserve special attention about the possible effects of environmental and anthropogenic changes on biodiversity (Dudgeon *et al.*, 2006; Kolding *et al*., 2012, Grownaris *et al*., 2018). The chemical composition in closed lakes is thus a function of hydrological regime and the geological and agronomic nature of the surrounding catchment area (Schindler, 1978). Biological production in aquatic ecosystems (both open and closed water bodies) is partly dictated by abiotic factors such as lake morphology, water volumes, and nutrients (Kolding and van Zwieten, 2012). According to Schindler (1978), external nutrient loading mostly explains the changes in phytoplankton production irrespective pattern and/or climatic seasonality in lakes. In tropical lakes, little seasonal changes and high temperatures lead to temporal biological variability and cause seasonal variations in water volumes with effects on nutrient dynamics (Lowe-McConnell, 1979).

The changes in lake water levels are a major indicator of the variations in the hydrological regime of the lake drainage basin related to variations in the runoff properties (land-use changes) and/or climatic changes as well as meteorological changes in the catchment (Vuglinskiy, 2009). The variability in the water budget components results in water volume changes, flooding, submergence of littoral communities, and changes in ecological processes in the lake (Gownaris, 2015). Water level fluctuations (WLFs) in aquatic systems affect their physico-chemical properties, assemblage structures, ecosystem functions, and ecological services (Coops and Hosper, 2002). The reduction in precipitation and increase in temperatures especially in arid and semi-arid areas leads to elevated water temperatures, high evapotranspiration rate, and reduced lake level and/or volume (Kolding and Van Zwieten, 2012; Okech *et al*., 2019; Herrnegger *et al*., 2021). Water level changes in lakes can be caused by natural factors such as climatic forcing (Bergonzini *et al*., 2004; Dudgeon *et al*., 2006; Gownaris *et al*., 2015), human influences such as through damming abstraction (Aloo, 2002; Evtimova and Donohue, 2014). The changes in lake level may in turn affect physico-chemical parameters through volume changes and productivity of the system through ecohydrological influences (Zalewski *et al*., 1997; Coops and Hosper, 2002). Changes in water levels (WLFs) tend to affect lake productivity through nutrient supply variations, influence on breeding areas, and changes in shallow productive inshore areas (Kolding and van Zwieten, 2012; Gownaris *et al*., 2015; Musinguzi *et al*., 2019) in addition to changes in water quality parameters (White *et al*., 2008). The influence of water level variations on water quality and fisheries production has been widely studied in most temperate rivers, lakes, and reservoirs (Welcomme, 1970;

Hamerlynck *et al*., 2011; Kolding *et al*., 2016). However, there has been a paucity of studies in tropical freshwater bodies and especially the Afrotropical systems.

East African Rift Valley Lakes, such as Lake Baringo, are sensitive to climate changes due to their arid/semi-arid location, shallowness, and water level variations in response to environmental changes (Ngaira, 2006; Gownaris, 2015; Okech *et al*., 2019; Herrnegger *et al*., 2021). Historically, the Kenyan Rift Valley region has been receiving an insufficient amount of variable annual rainfall (Ngaira, 2006, Okech *et al*., 2019) leading to no equilibrium between the water budget components of the lakes (Herrnegger *et al*., 2021). This has dramatically affected water levels of the Rift Valley Lakes, water quality, and fishery production (Gownaris *et al*., 2015; Okech *et al*., 2019; Herrnegger *et al*., 2021).

Lake Baringo in the Rift Valley of Kenya has had variable lake level changes due to droughts and floods (Odada *et al*., 2006; Aura *et al*., 2020; Herrnegger *et al*., 2021). The lake supports the local economy and livelihoods through fisheries and several economic uses of water (Odada *et al.*, 2006; Aura *et al*., 2020). However, all the benefits and multiple activities in the lake and its surroundings have increasingly put it under threats (Onyando *et al*., 2005; Omondi *et al*., 2014). The population surrounding the lake has been growing rapidly in recent years (KNBS, 2015). This has led to an increase in urban runoff, and effluent discharges from industries, agriculture, and mining activities in the lake catchment with potential effects on water quality (Aloo, 2002). The lake is thought to be experiencing eutrophication due to phosphate and nitrogen from intensive agricultural activities in its catchement (Aloo, 2002; Odada *et al*., 2006; Omondi *et al*., 2014). Lake Baringo is characterized by low water depths (average of 9.5 m, as in 2020, Walumona, personal observation), the mixing of surface and bottom water induced by the wave actions together with soils with limited vegetative cover in the lake basin contributes to the lake's notable high turbidity, reported to affect primary productivity of the lake (Odada *et al.*, 2006; Hinckley *et al*., 2014; Nyakeya *et al.*, 2020).

The lake's fishery was mostly based on the endemic *Oreochromis niloticus baringioensis* before the introduction of the marbled lungfish, *Protopterus aethiopicus* (Mlewa, 2003; Mlewa *et al*., 2006; Odada *et al.,* 2006). The fish landings have decreased in recent years due to overfishing and the cascading effects of environmental changes (Omondi *et al*., 2014). Also recently, some of the species such as *Labeobarbus intermedius (*Baringo barb*)* and *Labeo cylindricus* have not appeared or have appeared sporadically in fishermen's catches as they are becoming very rare in the lake (Aloo, 2002; Odada *et al.,* 2006; Omondi *et al*., 2014) and are likely functionally extinct. The fisheries of the lake could be affected by both overfishing, polluted water from its tributaries, and runoff which causes water quality degradation. Additionally, the water level fluctuations in Lake Baringo believed to be influenced by both climate change and human activities (deforestation, damming on rivers, and land use patterns) in its basin likely have influences on the lake's ecohydrology but have not been well studied. The extent to which water level fluctuations affect the lake's physico-chemical parameters and hence productivity has not been quantified but may be significant and could as well be related to fluctuations in fish catches and the loss of biodiversity in the lake.

For the sustainable management of the lake ecosystems; information on factors affecting fisheries dynamics, the water quality changes of the lake, the quantities and temporal variations of river inflows and influence on water balance, and the influences of water level fluctuations of the lake is important for intervention policies and management. This study therefore aimed at generating these sets of information for Lake Baringo, a rift valley lake in Kenya, for purposes of sustainable management of the ecosystem and the livelihoods of riparian communities.

1.2 Statement of the problem

The effects of climate change on African water systems are seen through interchanges in the hydrological cycle and the temperature (Lundgren *et al*., 2019). The lake level changes may expose or submerge littoral communities, and water volume changes may result in changes in physico-chemical properties, leading to changes in habitat quality for aquatic species which may, in turn, affect the distribution and abundance of fisheries species (Gownaris, 2015; Gownaris *et al.*, 2017). Reduced precipitation and higher temperatures can result in increased concentrations of pollutants in a lake due to reducing lake level and/or volume and warming surface water leading to high evapotranspiration (Kolding and Van Zwieten, 2012; Gownaris, 2015).

Warming of lake waters has been observed in tropical lakes in general including Lake Baringo (Hulme *et al*. 2001; Omondi *et al*., 2014). Lake level induced by water warming may affect primary and secondary production and the spatial distribution of fishes (Grownaris *et al*., 2015; 2017). However, there has been little quantification of these effects in African lakes (Kolding and Van Zwieten, 2012). Tropical African lakes are sensitive to climate change as their water balances are mostly dominated by direct precipitation on the lakes and evaporation, with river inflow and outflow making little contribution to the water balance (O'Reilly *et al*., 2004; Verberg *et al*., 2003; UNEP, 2004; Muvundja *et al.*, 2014; Grownaris, 2015).

Lake Baringo, a Ramsar site, is a shallow lake with high turbidity and high net evaporation that characterizes the Kenyan Rift Valley Lakes (Odada *et al*., 2006; Omondi *et al*., 2014). Lake Baringo basin is one of the worst-eroded areas in Kenya (Thom and Martin, 1983; Aloo, 2002; Odada *et al.*, 2006). The lake is also characterized by fluctuations in water levels due to climate change effects, dam construction, abstraction of water for irrigation and drinking purposes, and catchment deforestation, amongst others (Onyandi, 2005; Omondi *et al.*, 2011). Water level fluctuation (WLFs) affects productivity and influences the timing and breeding habitat areas and other biological events in aquatic systems (Kolding and Van Zwieten, 2012). Recently, there has been remarkable variations in climatic patterns (floods and droughts) that have negatively affected Lake Baringo water levels (Ngaira, 2006; Omondi *et al*., 2014; Aura *et al.*, 2020; Hermegger *et al*., 2021). However, the influences of water level fluctuations on the lake's physico-chemical properties and ecological functioning have not been well documented (see Walumona *et al.*, 2021a). In the tropics, rainfall remains the main factor that affects the aquatic ecosystems in arid and semi-arid areas such as Lake Baringo (Odada *et al*., 2006; Omondi *et al*., 2014). Water level fluctuations can alter habitat availability, complexity and quality depending on the morphology of the ecosystem and may lead to large variations in littoral habitats (Kolding and Van Zwieten, 2006; Kolding and Van Zwieten, 2016; Gownaris *et al*., 2017; Kolding and Van Zwieten, 2018) which is, for most lakes, considered as a reproduction habitat for most fish. It is likely that fish habitats in Lake Baringo may have been affected by changing water levels and hence recruitment rates and long-term fisheries harvests could be affected. The lake is known to be highly turbid due to volcanic soil deposited from erosion in the catchment (Onyando *et al*., 2005). Turbidity of the lake is likely to vary with lake levels with consequent variations in lake productivity and effects on ecosystem functioning and structure (Gownaris *et al*., 2018).

Recent and continuous time series in lake levels, river discharges, and meteorological data are important for updating and strengthening the models of lake water balance for flood forecasts and sensitivity analysis of Lake Baringo to climate variability (Aura *et al*., 2020; Hermegger *et al*., 2021). Therefore, many aspects of the functioning of the lake need to be examined to better understand the lake's responses to the environmental and anthropogenic drivers. The necessary data comprise, but are not restricted to, the sedimentation rate of the lake, the frequency of the fluctuations in water levels, the dynamics of the fish biomass and abundance, the biomass-trophic structure of the lake, the internal and external nutrient sources and the productivity and producer community structure (Schindler, 1978). Nutrient and water balance models of the lake are also needed for the assessment of the lake basin water and nutrient budgets for the establishment of the lake's responses to climatic and anthropogenic stressors.

This study, therefore, aimed to bridge the information gap with respect to the above parameters necessary for interpreting and predicting future limnological patterns and changes of the lake. The data are further necessary for the sustainable management of the lake's ecosystem. The study models lake level changes, the water balance and the tropho-dynamics of the lake.

1.3 Justification

Limnological studies are fundamental in understanding how physical, chemical, and biological processes interact and influence fisheries production in relation to the fluctuating water levels of lakes (Kolding and Van Zwieten, 2012). Water level variations affect the abundance and species composition, stability, productivity, and physiological condition of indigenous populations of aquatic organisms (APHA, 2005). Very little is known about how water level changes affect the ecological functioning of African lakes (Kolding and Van Zwieten, 2012). Lake Baringo is one of the Kenyan rift valley lakes where the water levels have been fluctuating overtime at different temporal scales. These changes have been related to natural events (climate change) and human activities (damming of river inflows and land use) in the lake catchment (Odada *et al*., 2006; Aura *et al.*, 2020). These changes have recently been observed but are not known to affect the fish species composition and production from the lake (Walumona *et al*., 2021b) and may also influence fluctuating trophic status of the lake over time perhaps contributing to the declining fishery of the lake (Odada *et al*., 2006; Omongi *et al*., 2014; Walumona *et al*., 2021b).

Studies conducted in Lake Baringo have focused on fisheries production of the dominant and endemic *Oreochromis niloticus baringoensis* and its ecology (Odada *et al*., 2006; Omondi *et al.*, 2011; Omondi *et al.*, 2014; 2016; Nyakeya *et al.*, 2020), the ecology of *Protopterus aethiopicus* (Mlewa, 2003; Mlewa *et al*., 2006) amongst others. Although the decline in fish species catches is mostly attributed to overfishing (Aloo, 2002; Odada *et al*., 2006), the changes may, in part, be attributed to changes in water quality and habitat structure due to the fluctuating lake levels but there have not been studies to relate cause and effect of WLFs in the lake. An integrated approach that models fish catches in relation to ecosystem functioning such as Ecopath with Ecosim (EwE) modeling (Polovina, 1984, Christensen, 1995) is necessary to determine the link between fisheries and the ecosystem. To accomplish this, data on water level fluctuations and the water balance model of the lake is needed in the assessment of the lake basin water budget for the establishment of the lake's sensitivity to the climatic and anthropogenic stressors.

Limnological, hydrological, and food web models (such as Ecopath with Ecosim, (EwE), Christensen, 1995) are necessary for a better understanding of the changes in the lake water balance, nutrient components, and the interconnection between the trophic levels in the lake's food web, water quality, and lake water levels. The modelling and forecasting of future limnological and ecological changes of Lake Baringo are important for sustainable management of the lake to ensure ecological stability and continued provision of livelihoods to riparian communities.

1.4 Objectives of the study

1.4.1 Main Objective

The main objective of the research was to study the effects of recent and long-term water level fluctuations on the ecohydrology, and fishery of Lake Baringo in addition to modelling the water balance and trophic relationships of the lake.

1.4.2 Specific objectives

The specific objectives of the study were:

1. To determine the spatio-temporal variations in water quality and trophic status of Lake Baringo at intra- and inter-annual temporal scales**.**

2. To model the water balance of Lake Baringo and its sensitivity to variations in hydro-meteorological variables.

3. To evaluate the effects of lake level changes on the water quality variables and the fisheries yields (landings) at intra- and inter-annual temporal scales.

4. To model the ecosystem functioning in Lake Baringo using the Ecopath mass-balanced model using food web relationships and energy flow vectors.

1.5 Hypotheses

This study was guided by the following null hypotheses:

H_{01}: The water quality and trophic status of Lake Baringo vary at spatial and intra- and inter-annual temporal scales.

H_{02}: Changes in hydro-meteorological variables influence the water balance components of Lake Baringo and the lake is sensitive to variations in hydro-meteorological variables.

H_{03}: Lake Baringo water level fluctuations affect the fishery and water quality parameters of the lake at intra- and inter-annual temporal scales.

H_{04}: The lake's ecosystem functioning and structure have been unstable over time based on the energy flow between the different functional groups in its food web.

1.6 Thesis structure

The thesis is organized as follows:

The thesis chapters are preceded by an abstract that summarizes the findings of the study.

Chapter 1 summarizes the background information on lake functioning, the effects of water level changes on the fisheries, and water quality in African lakes including Lake Baringo. It also presents the statement of the problem and justification of the study, gives the objectives that guided the research, and presents the hypotheses of the study.

Chapter 2 contains an extensive literature review and summarizes the physico-chemical properties of the lake, the ecosystem services provided by certain lakes, and possible threats to the lake's functioning. The relationships between the physico-chemical properties and the lake's responses are discussed as well as the impacts of climatic variability on the lake. The lake's fisheries, composition, importance, and threats are also discussed. There is a brief review of the Ecopath with Ecosim (EwE) model used to study the lake's food web.

Chapter 3 presents the different materials and methods used to address the objectives of the thesis. It also gives details on the sampling frequency, types of equipment, and techniques used for the analysis of water quality parameters, water level measurements, and fisheries data collected from the lake. It provides the statistical analyses and applications used in the present study. The methods are arranged sequentially according to the objectives.

Chapter 4 presents a description of the results of the study based on the objectives and applied methods.

Chapter 5 discusses the results presented in Chapter 4.

Chapter 6 contains the major conclusions and recommendations of the study based on the objectives.

References are listed at the end of Chapter 6, while Appendices are found after the references.

The thesis structure follows the guidelines provided by the University of Eldoret for writing theses.

CHAPTER TWO

LITERATURE REVIEW

2.1 Overview of African Lakes

2.1.1 Location and origin of African Rift Valley Lakes

The large East African Rift Valley Lakes (EARVL), to which Lake Baringo comprise, extend from the northern end of Lake Turkana Basin to the southern tip of Lake Malawi/Nyasa Basin and include all the natural habitat and associated human populations along the Rift Valley and on the adjacent escarpments (Figure 2.1). It comprises parts of the following countries; Kenya, Ethiopia, Sudan, Uganda, Tanzania, Rwanda, Burundi, Democratic Republic of Congo (DR. Congo), Zambia, Malawi, and Mozambique. The main lakes in the ecoregion are tropical and include the African Great Lakes: Turkana, Victoria, Tanganyika, Edward, Albert, Kivu, George, and Malawi. Each lake is different from others regarding the limnology, an assemblage of endemic organisms, catchment dynamics, drainage basin characteristics, and human influences (Hamilton, 1982). The lakes of the East African Rift Valley (Figure 2.1) constitute unique natural resources and play important roles in the riparian countries in the tropic region. The lakes, therefore, contribute to climate moderation, transportation, water supply, fisheries, waste disposal, recreation, and tourism. These services attract huge populations around lakes leading to negative influences (Cohen *et al*., 1996).

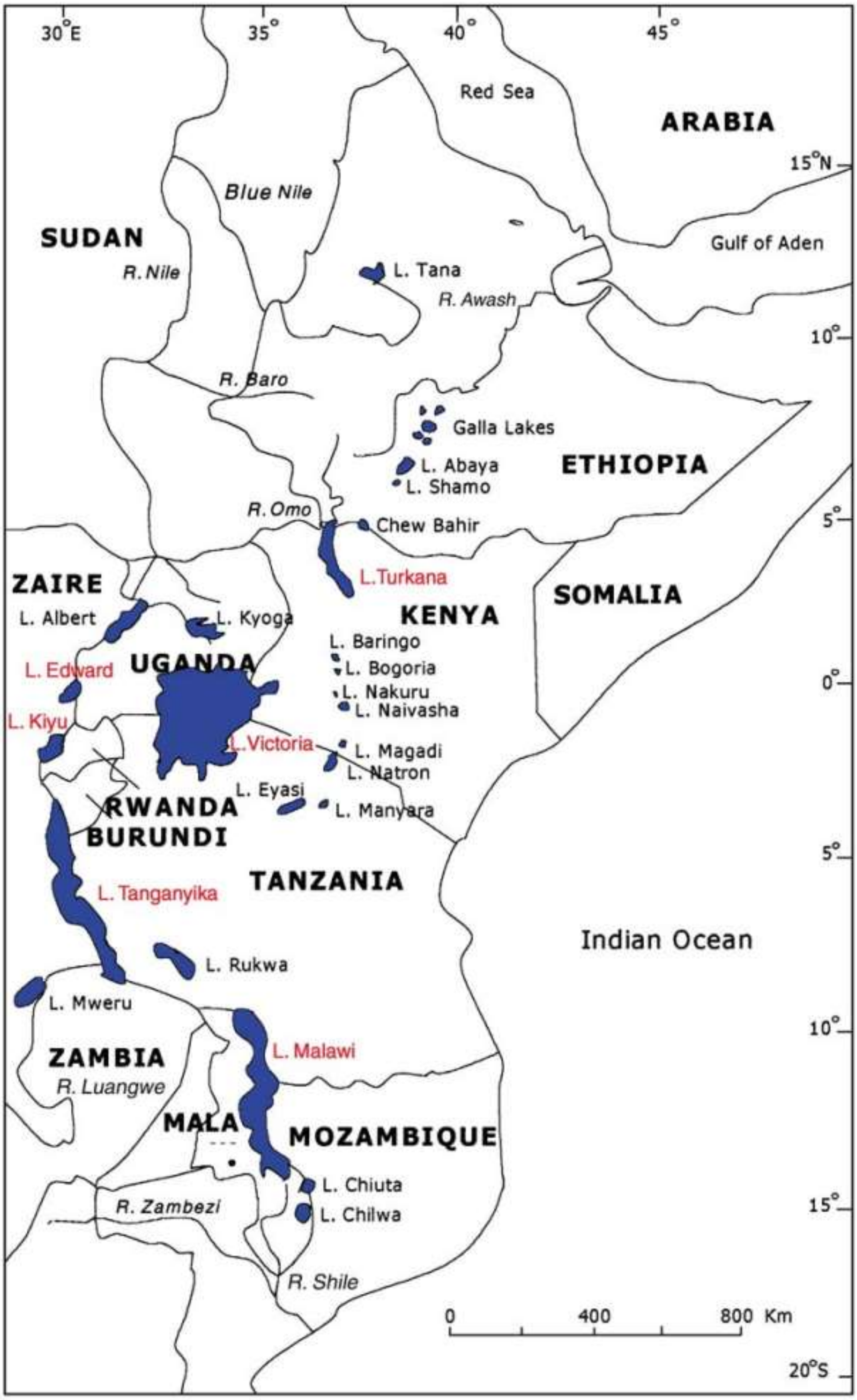

Figure 2.1: Map of Eastern African lakes showing their neighboring landmasses. Adopted from Nyamweru (1983).

Twelve million years ago, a tectonic fracture occurred on the African continent rising the Red Sea and a large part of the East African Lakes (Hamilton, 1982). The lakes of East Africa including Lake Baringo (Figure 2.2) were born from this fracture, some by filling pools created by west and east cleft formations like Lake Victoria and others by filling in the gaps created like Lakes Tanganyika and Malawi. These African lakes have an exceptional long geological existence, an uncommon behavior of lacustrine ecosystems (Wetzel, 1983). The Kenyan Rift Valley Lakes originate from volcanic activities with ongoing seismicity and geological exploration discovered late 19th century (Schlueter, 1997). Major differences in Rift Valley lakes in Kenya are seen in the dissolved salts ranging from the large alkaline Lake Turkana in Northern Kenya to small and shallow freshwater lakes such as Lakes Baringo and Naivasha, to saline or brackish Lake Nakuru and hypersaline Lake Bogoria in Central Kenya (Campbell *et al*., 2003).

In the following section a description of the functional literature is made relating to the objectives of the thesis and highlighting the information gaps and provides a rationalization of the study.

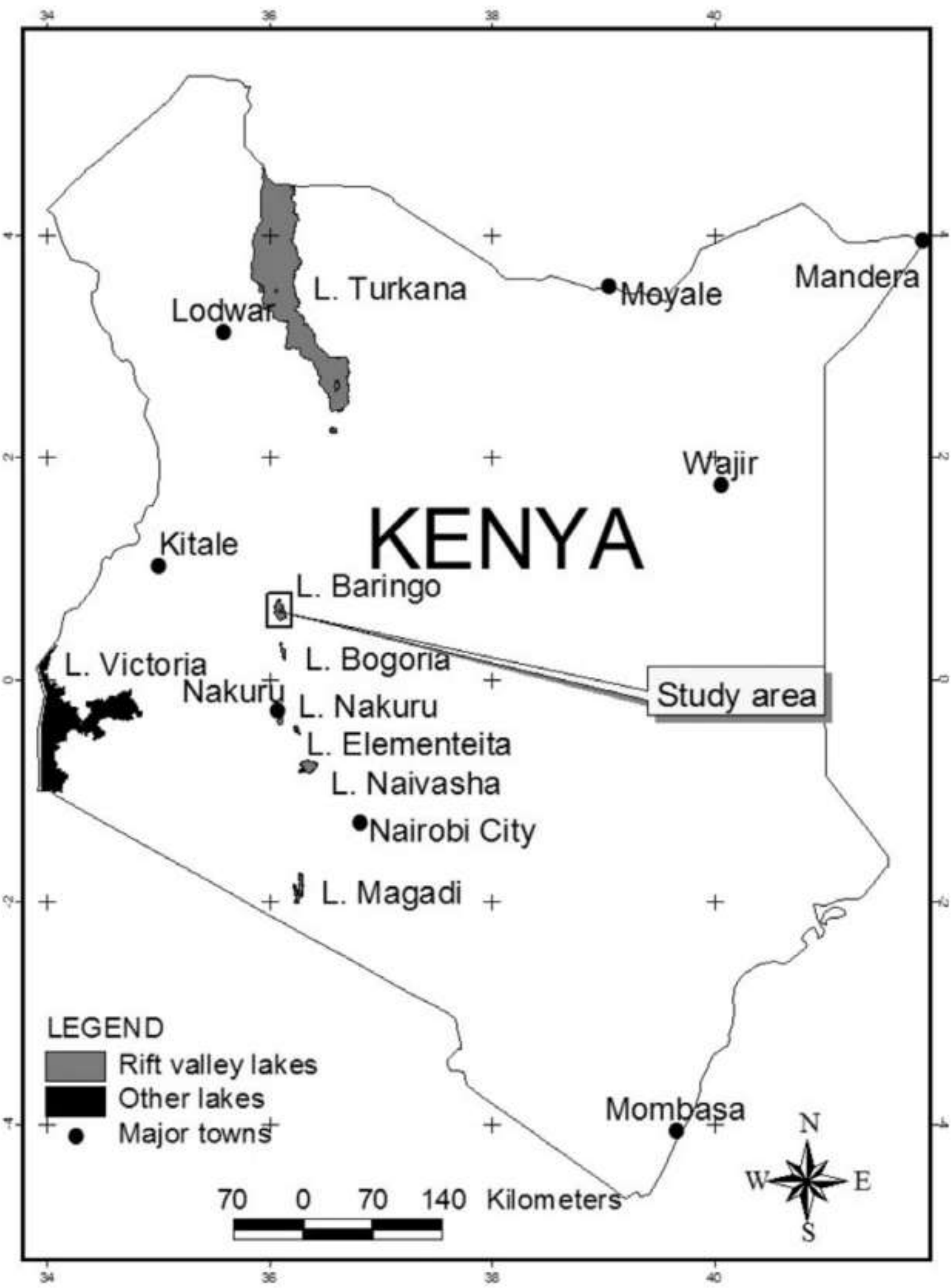

Figure 2.2: Map showing the location of lakes in the eastern arm of the Kenyan Great Rift Valley including Lake Baringo, adopted from Nyamweya et al. (2010).

2.2 Water Quality impacts and Trophic Status in Lakes

The quality of water in an ecosystem (lakes or rivers) reflects the nature of the environment, the water body properties, and the ongoing anthropogenic activities in the ecosystem's watershed over time (Kithiia, 2011). Human activities are considered as environmental stressors that may accelerate natural processes affecting the water body's physico-chemical characteristics (Yidanaa *et al*., 2008). Rapid urbanization has been reported to make the riparian land to be covered by impervious surfaces that facilitate increasing surface runoff and erosion of river and stream beds towards lakes (Obire and Aguda, 2002). The urban storm runoff contributes to the pollution of lake ecosystems by loading nutrients, fecal microbes, and metals into the water especially for Lake Baringo (Aloo, 2002). Agricultural and industrial activities cause land-use changes resulting in the introduction of various harmful pollutants such as suspended sediments, heavy metals, organic and inorganic nutrients due to fertilization practices, fecal microbes, pesticides, and herbicides responsible of water pollution and high turbidity levels in the lakes (Ahuja, 2009). High turbidity levels in the lake water are known as the oxygen depletion factor in the ecosystem by limiting the photosynthesis process (Karenge and Kolding, 1995; Teng *et al*., 2007). Oxygen depletion depends on the total amount and nature of the organic material load in the lake and makes the aquatic life conditions stressful for some organisms in the ecosystem (Mason, 2002).

High soil erosion rate caused by overgrazing due to livestock and deforestation in the watershed of the lake results in the siltation in the lake that may affect some species communities and primary production of the lake (Chapman *et al*., 1992). Siltation has been reported to have two sources; natural and human, and contributes to lake level fluctuations (Olilo, 1993) including in Lake Baringo (Aloo, 2002). Besides, the damming of rivers inflowing into lakes and abstraction from lakes for irrigation and drinking purposes affect the lakes' water quality and ecological functioning by fluctuating their water levels thus affecting the feeding habitats and refugee areas for some species (Chapman *et al*., 1992; Ssentongo, 1996).

Natural processes as well as exogenic influences play a big role in surface water degradation and weaken their agricultural, industrial, drinking utilizations as well as other purposes (Carpenter *et al*., 1998; Kazi *et al*., 2009). In addition, anthropogenic activities are considered as the major source of nutrients and trace metals which contaminate water bodies and make water quality poor for aquatic life (Sondergaard *et al*., 2003; Li *et al*., 2009). In most cases, nutrients such as nitrogen, phosphorus and carbon coming from human activities entering surface waters through wastewater and agricultural uses are known as the major source of water eutrophication in lakes (Kotak *et al.* 2000; Downing *et al*., 2001; Oboh and Agbala, 2017).

Considering that human and natural activities cause many problems to ecosystems, lakes and rivers deserve to be given more attention since their waters properties can get altered and threatened by anthropogenic activities (Makhoukh *et al.*, 2011). Despite the many influences of riparian activities on lakes affecting water quality, it is not known how water quality changes affect lake functioning in many Afroafrican systems (Downing *et al*., 2001; Aloo, 2002). Therefore, the understanding of interaction of water with lithologic units through which it flows is important for water chemistry and quality controls (Subramani *et al*., 2009). The lake watershed interractions determines the nature of setting of geochemical properties of water which is an important factor determining its use for different purposes **(**Giridharan *et* *al.*, 2010**).** Consequently, water monitoring program of aquatic ecosystems (Lakes and rivers) seems to be of vital importance t to establish whether the lake water properties are admissible for aquatic life and/or various uses (Pesce and Wunderlin, 2000; Sener *et al*., 2017).

Several approaches have been developed and used to assess the water chemistry and status of water quality in rivers, reservoirs and lakes (Aston *et al*., 1980; Afsin, 1997; Yidana and Yidana, 2010; Sener *et al*., 2017). Thus both water quality Index (WQI) and organic pollution Index (OPI) have been used to evaluate water quality of lakes and rivers both inorganically and organically (Leclercq and Maquet, 1987; Meybeck and Helmer, 1989). Studies by Ramakrishnaiah *et al*., (2009), and Sehner *et al*., (2017) assessed water quality of Tumkur and Karnataka (India) and of Aksu River (SW-Turkey) using Turkish water quality index but this is not as universal as the internationally accepted WQI or the OPI. Also, Yidana and Yidana (2010) used conventional graphical methods based on multivariate statistical methods and GIS to evaluate the controls on the hydrochemistry and the integrity of the controlling factors at different locations in a flow system. Furthermore, water quality index (WQI) method has been used to assess the suitability of groundwater for human consumption and environmental suitability (Sener *et al*., 2017). Kannel *et al*. (2007) have also used WQI to assess spatial and seasonal changes in the water quality in the Bagmati river basin. Other authors like Debels *et al.* (2005) calculated WQI in order to characterize the spatial and temporal variability of surface water quality in the basin, from nine physico-chemical variables. WQI is defined as a rating tool reflecting the composite effect of various water quality variables (Sahu and Sikdar, 2008). Thus, the WQI has been considered as criterion for surface water classification based on the use of standard parameters for water characterization (Sener *et al*., 2017). It also provides a comprehensive picture of the quality of water for domestic usages, facilitates a water quality comparison among several sites chosen along a lake or a river and constitutes an easier mathematic tool which transforms great amounts of water characterization data to a simpler number that represents the water quality level (Bardalo *et al.*, 2006) and quantifies an individual effect of each and every parameter on the Lake water quality (Yidana and Yidana, 2010). In this study, WQI and IPO are used for the first time in Kenyan aquatic systems to describe temporal and spatial variations in the quality of Lake Baringo water and recommend use of the water to support livelihoods.

The trophic status index is a universal measure of water quality and eutrophication stress (Carlson, 1977). Studies on the trophic status variation in lakes have demonstrated that eutrophication is slow and a long process but it is one of the greatest risks to aquatic ecosystems (Zbierska *et al*., 2015). Nutrient loading from the catchment has been reported to be responsible of phytoplankton growth over a long period of time and can also build up in the sediment creating the potential for an internal load facilitating the resuspension into the water column under different environmental conditions (Hou *et al*., 2013). Chlorophyll-a is used to estimate the amount of phytoplankton or algae in a water body. It is a useful tool for determining the biological productivity of an aquatic system (Mahesh *et al*., 2014). It is therefore an accurate parameter used for prediction productivity and algal biomass and evaluation of the trophic status of a water body attributed with the highest classification priority compared with other variables (Transparency, TP and TN) (Murphy et al., 2008). For trophic level assessment, some indicative variables such as, chlorophyll a, total phosphorus, transparency, and total nitrogen are taken into account (Carlson, 1977; Popovicova and Celi, 2009; Rahul *et al*., 2013).

Also, the Trophic State Index (TSI) introduced by Carlson (1977), an average of the above individual parameters, is widely used and acceptable index to estimate the limiting nutrient causing eutrophication (Nalamutt and Karmakar, 2014), and recommended for nutrient loading into lakes and reservoirs (EPA, 2000).

The trophic status of lakes is also evaluated using the nutrient availability considering the levels of readily bioavailable inorganic nutrients such as TN: TP ratio and SRP:DIN ratio (the latter is the sum of nitrate, nitrite and ammonium (OECD, 1982; Reynolds, 1999). The N limitation is considered probable when the molar TN:TP ratio is < 10, while P limitation when the TN:TP ratio is > 20 (Stephen *et al*., 2020).

The effects of the pollutants on the Lake Baringo water quality and trophic status at both short- and long-term scales are not well established. This study is therefore more important for the region and aimed at the assessment of the physico-chemical properties of the Lake water using the water quality indices (WQI) and OPI (organic pollution index), and determine the variable that influences individually the water quality of the lake. It also determined the trophic status of the lake at intra-and inter-annual scales.

2.3 Lake water Balance models and Hydro-metereological components

The effects of climate change are seen in the increase of water temperatures and evaporation in many lakes through risen air temperature, in both temperate and tropical regions (Zinyowera *et al*., 1998; Schindler, 2001). This is likely to lead to declines in inflows, outflow, and/or lake volume if the increase in rainfall does not compensate significantly. Tyedmers and Ward (2001) indicated that the lake water levels and characteristics might be affected by the warming temperature and high evaporation rate leading to the reduction and/or loss of fish habitat in the lakes. Also, Schindler (2001) demonstrated that freshwater lakes with outlets might become closed lakes as a result of high temperature and evaporation due to climate change without supplying the downstream systems leading to an increase of the salinity level in endorheic freshwater lakes. Variations in mean air temperatures have also been shown to raise water temperatures in both temperate and tropical lakes, affecting their physico-chemical and biological properties (Yin and Nicholson, 1998; Yin, *et al*., 2000; Verberg *et al*., 2003; Verschuren, 2003).

African Great Lakes and especially closed lakes are more sensitive to climatic changes (Verschuren, 2003, Okech *et al.*, 2019) attributed to the lack of surface outflow and over-dependence on river inflow for recharge and precipitation-evaporation control on their water budget (Spigel and Coulter, 1996; Bergonzini, 1998; Odada *et al*., 2003). However, the African Great Lakes' sensitivity to climate change is mostly related to the climatic effects on river input and/or outflow, the contributions to the lakes' water budget being very small, and their dependence on precipitation estimated at 80-90% for the survival of these basins **(**Vuglinskiy *et al*., 2009; Becker *et al*., 2010). Otherwise only minor declines in rainfall (10-20%) are expected to completely close these basins (Bootsma and Hecky, 1993) making it useful to determine the input and output variables of a lake. Most Great African Lakes basins such as Lakes Kivu, Zaway, Tanganyika, Victoria, and Malawi depend on the precipitation contributions more than the net inflow to their water balance (Bootsma and Hecky, 1993; Bergonzini, 1998; Vallet-Coulomb *et al*., 2001; Kumambala and Ervine, 2010; Muvundja *et al*., 2014). This study aimed to determine the relative dependence of Lake Baringo on the water balance parameters (rainfall, evaporation, seepage and runoff).

Compared to temperate lakes, tropical lakes don't display sharper thermal gradients and larger seasonal changes in water temperature due to their position and seasons with stable temperature, but their chemistry may be affected by climate change in different ways (Schindler *et al*., 1990). Drought conditions and decreasing groundwater inflow expose some lakes to acidification, by breaking the acid-neutralizing chemicals equilibrium in groundwater that is important to lake buffering (Schindler, 2001). Temperature and precipitation changes affect the chemical balance and water balance components in some water bodies in addition to their ecological processes in a number of ways (Verberg *et al*., 2003). White *et al*. (2008) found significant correlations with water quality parameters (DOC, Ca^{2+}, conductivity, pH, SO_4^{2-}) and water level fluctuations (WLFs) indicating patterns in natural WLFs and associated correlations with water quality in Boreal Shield research regions. Water balance parameters will, therefore, affect lake levels and their water chemistry. Human activities such as river damming have been reported to affect lake levels in Africa (Omondi *et al*., 2014; Grownaris, 2015). The water budgets on lake level changes are not known for most Afrotropical lakes including Lake Baringo but are well established for temperate lakes (White *et al*., 2008)**.** Additionally, the parameters that affect the water balance of African lakes are only known for a few lakes such as Lakes Malawi and Kivu among others (Kumambala and Ervine, 2010; Muvundja *et al*., 2014). Consequently, this thesis establishes the relative importance of water balance components in affecting the levels of Lake Baringo. Information is laking for other Kenyan lakes.

2.4 Effect of Lake Level Changes on Water Quality and Fish Yields

Although the effects of climate change on the biotic communities of tropical lakes have not been widely documented in general, very little attention has been given to tropical lakes in Africa (Coops *et al.*, 2003). Most studies demonstrate that a water level fluctuation (WLFs) between 1.5 and 2.0 m due to natural and/or human activities is the optimal level at which the highest macrophyte diversity is attained in lakes (Wilcox and Meeker, 1991; Hill *et al*., 1998; Wagner and Falter, 2002). However, large decreases in primary productivity due to climate warming effects on lake levels have been studied in Lake Tanganyika (Verberg *et al*., 2003) and are likely to have a significant impact on the rest of the food chain. The impacts of climate change are seen through the effects of water level variations and air temperature changes (Verschuren, 2003). Water levels directly affect emergent aquatic plant biomass (Wetzel, 1983) and have an indirect influence on submerged aquatic plant biomass through the decrease of light penetration intensity (Chambers and Kalff, 1986; Duarte and Kalff, 1990; Middelbøe and Markager, 1997). Some studies on relationships between water levels and biota indicated a significant relationship of macroinvertebrates species richness with natural WLFs demonstrating a partnership between these variables within the Laurentian Great Lakes region (White *et al*., 2008). Riis and Hawes (2002) found along 21 lakes in New Zealand the highest species richness of low growing mixed macrophyte community at 1.1 m fluctuation of water level. The same study indicated the importance of both inter-annual and intra-annual WLFs in establishing high levels of species richness.

Studies on Lake Turkana, one of the African rift valley lakes in Kenya, showed that water level fluctuations are the key drivers of fisheries productivity in the lake ecosystem (Grownaris *et al*., 2015). The same study indicated that the WLFs alter the distribution of habitat types and seasonal flood pulses affecting some fish species of economic importance that are ecologically flexible (e.g. *Oreochromis niloticus*, *Lates niloticus*) and others (e.g. *Tilapia zillii*, *Labeo horie*) that are highly sensitive to changes in habitat availability and food web structure. The results of the study on Laurentian Great Lakes region (White *et al*., 2008) highlighted also the sensitivity of macroinvertebrates to WLFs compared to macrophytes and fish species indicating the highly responsive nature of macroinvertebrates to WLFs and their usefulness in assessing the associated effects on water bodies. In Lake Chilwa in Malawi, a shallow lake, seasonal fluctuating water levels affect the availability of refuge areas (Allison *et al*., 2006; Wantzen *et al*., 2008a) and affect differently the fish species depending on the resilience of organisms to such changes (Welcomme *et al.*, 2010; FAO, 2012).

The influences of lake level changes on water quality parameters are not well established for most Afrotropical lakes including Lake Baringo but are well established for temperate lakes (White *et al*., 2008)**.** The information on the combined impacts of climate change and other effects on shallow-unstable lake system's fishery is limited in the literature. This study aimed to fill the knowledge gap with regard to how African lakes respond to WLFs using Lake Baringo as a test case.

2.5 Modelling of Ecosystem Functioning using Food Web and Energy Flow models – Ecopath with Ecosim (EwE)

2.5.1 Overview of Ecopath Model

Ecosystem models are largely used to study the interactions that occur within a system (Christensen, 1995), including those between different organisms and those between fisheries and targeted species (Ribeiro *et al.*, 2019). Trophic web models like Ecopath with Ecosim (EwE) (Christensen, 1995) can handle fishing fleets as a top predator, with a top-down impact on harvested organisms and energy flows between different functional groups in the food web. Ecopath with Ecosim model (EwE) is a descriptive, static, and dynamic model used to identify trophic interactions among functional groups within an aquatic system during a given period (Patterson and Kachinjika, 1995). The primordial use of EwE is the determination of energy budgets, trophic flow, and, area protection function and ecosytem structure (Christensen, 1995).

The Ecopath with Ecosim modeling package (Pauly *et al*., 2000) comprises three main components: (i) Ecopath is a static, mass-balanced snapshot of the system; (ii) Ecosim is a time dynamic simulation module to allow prediction of the response to system perturbation, such as through fishing exploitation, and; (iii) Ecospace is a spatial and temporal dynamic module designed primarily for exploring the impact and placement of protected areas. Ecopath, based on an approach developed by Polovina (1984) for the estimation of biomass and food consumption of the various species or groups of species of the food web, was combined by Christensen *et al.* (2005) with various approaches from theoretical ecology, notably those proposed by Ulanowicz

(1986), for the analysis of flows between elements. The core routine of the Ecopath with Ecosim (EwE) model is basically function of two master equations, one to describe the production term, and one for the energy balance of each group (Christensen, 1995). This ecological tool has been largely used for quantitative descriptions of aquatic systems and the evaluation of fishing impacts in water bodies (Christensen and Pauly, 1993; Christensen and Walters, 2004a).

Ecopath with Ecosim modeling tool originates from classic ecology. Food webs are ecologically based on trophic flows between discrete trophic levels comprising functional groups (Lindeman, 1942) and the latter (species) occupy distinct trophic levels and positions in a food web.

The Ecopath model described here presents, routines of estimating biomass, or production/biomass ratios, as well as food consumption by the various elements of a steady-state trophic model and routines based on the theory of Ulonowicz (1986) for analyzing the flows estimated by applying the first routines to the data. Additionally, this part of Ecopath with Ecosim model has routines for deriving further statistics from the biomasses and flows based on the above routines. It also attempts to quantify a number of Odum's (1969) 24 indices of system maturity.

Although this software is a free online progressing software and is constantly updated, the principles have not changed (Christensen, 1995; Christensen *et al*., 2005). The Ecopath model represents an ecosystem in which there is an interconnection between trophic groups based on biomass and is linked by mass transfers (Christensen and Pauly, 1993).

The Ecopath model, part of EwE, is used in this thesis in the assessment of the ecosystem functioning of Lake Baringo including fisheries impacts. The model has been commonly applied in temperate regions on more than 400 water bodies including rivers, reservoirs, oceans, and lakes to assess the energy flows, trophic structure, and system functions (Christensen *et al*., 2000; Pauly *et al*., 2000; Christensen and Walters, 2004; Xu, *et al*., 2011; Ullah, 2012; Colléter *et al*., 2015). Only a few studies using EwE model have been conducted in tropics to evaluate the ecosystem functioning in both lakes and reservoirs. Lakes such as Victoria (Tanzanian part) (Moreau, 1995), Kariba in Zimbabwe (Moreau 1997), Bagre Reservoir in Burkina Faso (Villanueva *et al*., 2006), Tanganyika (Tanzanian part) (Sarvalla *et al*., 1999), Malawi (Malawian part) (Darwalla, *et al*., 2010), Awassa in Ethiopia (Fetahi and Mengistou, 2007) and Volta in Ghana (Mensah *et al*., 2019) have been studied using the Ecopath model to assess the trophic interactions and temporal dynamics of their fish species for the sustainable management of their fisheries. In Lake Victoria, for example, the Ecopath model was also used to reflect the ecosystem state in Winam Gulf in Kenya and to construct a new model for the entire lake following earlier models. The modellling highlighted that exploitation is unbalanced and skewed to the least productive species at high trophic levels with less fishing effects at lower productive trophic levels (Natugonza *et al*., 2016). Other Ecopath models have been applied to study the functioning of African lakes (Darwalla, *et al*., 2010; Villanueva *et al*., 2006). This thesis describes results of Ecopath modeling, for the first time, for Lake Baringo and only the second such study after Lake Victoria (Natugonza *et al*., 2016).

CHAPTER THREE

MATERIALS AND METHODS

3.1 Description of Study Area

This sub-section provides in addition to the lake description, the general morphological and ecohydrological information of the study area. It also presents information based on secondary data on some physical, biological and chemical parameters of the lake measured as well as on lake level collected previously in order to provide a basic understanding of the limnology and ecohydrology of the lake.
The methods used to collect data under each of the objectives of the study are described below in the next sub-sections.

This study was conducted in Lake Baringo (Figure 3.1); a shallow lake in the eastern arm of the Great Rift Valley in Kenya (Beadle, 1932). Its commercial fishery depends on the naturalized population of the marbled lungfish *Protopterus aethiopicus* that was introduced in the lake in 1975 (Mlewa, 2003). The lake is also a designated Ramsar site, famous for its high bird diversity, hippopotamus and crocodile populations (Odada *et al*., 2006).

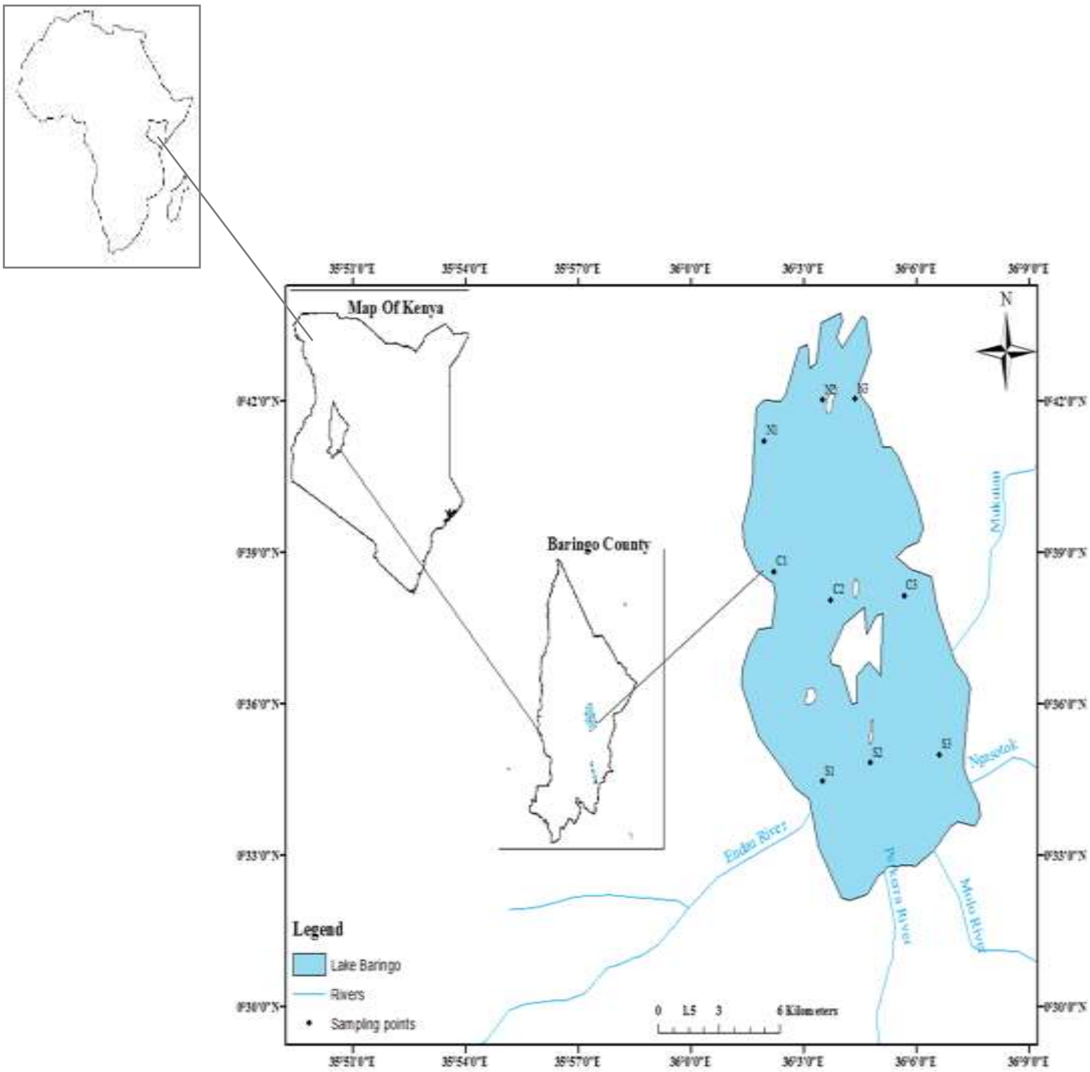

Figure 3.1: Map of Lake Baringo (Kenya) showing the sampling stations (S2 = southern station; C1, C2, and C3 = three stations in the central part and N2 = northern station) (adopted from Nyakeya et al., 2020).

3.1.1 Origin, and evolution

Lake Baringo is considered as the remnant of a larger freshwater with high salinity and alkaline lake known as Lake Kapthurin, which was developed in the axial graben during the Pleistocene era (Renaut *et al*., 1999, 2000). This lake is a small and shallow perennial freshwater lake that lies between latitude 0°30′N and 0°45′N and longitude 36°00′E and 36°10′E in the axial graben of the central Kenya Rift (Tarits *et al*., 2006), lying approximately 60-km north of the Equator at an altitude of 975 m above sea level (Kallqvist, 1987). The lake is reported to have a surface area estimated at 130 km^2 and a catchment of 6,820 km^2 (Ondiba *et al*., 2018) with an average depth of 3 m, and the deepest point being about 7 m (Odada *et al.*, 2006). Geophysical evidence has shown that modern Lake Baringo and the Loboi Plain are the surface expression of an 8 km deep, fault-controlled basin that was initiated during Palaeogene times (Hautot *et al*., 2000). The northern and central parts of the lake have several small fault-controlled islands, the largest island being the Ol Kokwe. The latter is the remnant of a small volcano that belongs petrogenetically to the Korosi volcanic massif and erupted during the Middle Pleistocene times (Clement *et al*., 2003), while the smaller ones are often submerged during periods of high water. The Soro hydrothermal system, situated in the northeastern part of Ol Kokwe Island, is clearly linked to the former volcanic activity (Tiercelin *et al*., 1987; Renaut *et al*., 2002).

3.1.2 Geomorphology

The geology of the area is mainly undifferentiated volcanic rocks, while the soils are of clay type (Ballot *et al*., 2003). The lake occupies an area characterized by active tectonics, recent volcanism, and high sedimentation rates in an intensively faulted area (Tiercelin *et al*., 1987; Hackman, 1988; Dunkley *et al*., 1993). The lake is reported to be 21 km long and 13 km wide with < 4 m average depth, lying in the axial graben of the Kenya Rift at an average altitude of 975 m (Odada *et al*., 2006). The maximum elevation of the watershed is about 2500 m (Tarits *et al*., 2006). The lake is bordered by littoral marshes and peripheral mudflats that facilitate the lake-ward extension for hundreds of meters during periods of low lake level, except along the northern and mid-western shorelines (Renaut *et al*., 2000; Owen *et al*., 2004).

3.1.3 Human population around the lake

The human population around the lake depends on the lake for socio-economic services such as water for domestic use, fishing, livestock, and agriculture and has been estimated at between 8,000 and 10,000 people (KNBS, 2015). The community especially the youth depends on this lake as the only source of income or employment. This is for example the case of beach boys who depend on tourists that visit the lake, and the fishermen who rely mainly on the lake as their only source of income. The main fishing village on the shores of Lake Baringo (Kampi ya Samaki) has a population of about 1,500 people (KNBS, 2019). Most of the riparian community depends on the lake for their daily livelihood as a source of food and water for domestic use. The lake also serves as a watering point for livestock.

3.1.4 Changes in water levels

The water level of Lake Baringo has been fluctuating over time, mainly in response to short and long-term hydro-meteorological changes and as a function of the precipitation/evaporation balance (Herrnegger *et al*., 2021). Both El Nino and La Nina climatic events have been recognized in the region (Johansson and Svensson, 2002; Hickley *et al*., 2003**;** Ashley *et al*., 2004), and these have controlled some of the larger fluctuations of lake levels. Between 1950 and 1979, lake levels fluctuated between ~ 965 m and ~ 973 m above sea level (Renaut and Owen, 1980). Since 1978, there has been a general decline in lake level, punctuated by brief periods of high levels in 1997–1998 (Hickley *et al*., 2003, Figure 3.2). The lake was 8.6 m deep in 1975 but reduced to about 2.1 m deep in 2001 (Johansson and Svensson, 2002). The deepest waters lie in the northwestern zones of the lake whereas the southern zone is the shallowest part of the lake (Tiercelin *et al*., 1987; Hickley *et al*., 2003) (Figure 3.2a). However, low lake levels and shoreline retreat were reported since 2000 and had raised concern that the lake was eventually drying up to become a swamp, particularly if water continued to be withdrawn for irrigation in the catchment (Aloo, 2002). However, due to heavy rains experienced in 2011 in the Eastern African region, the lake water surface increased dramatically to 207 km^2 in 2016 (Obando *et al.,* 2016) with the deepest point estimated at 11.22 m in June 2017 (Nyakeya *et al*., 2018) and then to more than 250 km^2 in 2020 (Hermegger *et al*., 2021, Figure 3.2b), the deepest point was found to be 15.8 m in 2019 (Nyakeya *et al*., 2020). Despite the perennial changes in lake water levels, very little is known as to how these affect the water quality and ecological functioning of the lake.

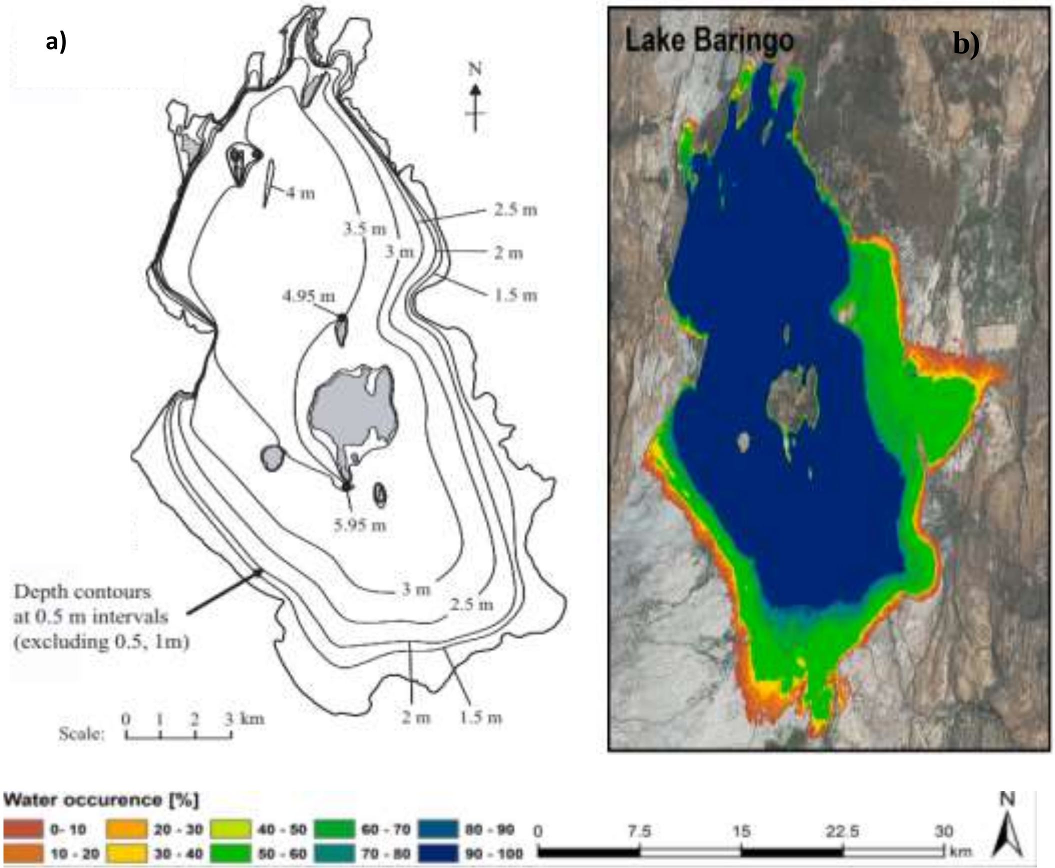

Figure 3.2: a) Bathymetric map for Lake Baringo (August 2003) (Hickley *et al*., 2003) and b) water occurrence from 1984 to 2020 in the lake emphasizing the variability of the surface area (Hermegger *et al*., 2021).

3.1.5 Historical changes in the limnological status

The physical conditions of Lake Baringo are characterized by high temperature and high turbidity levels that arose partly due to the high suspended matter from groundwater (Odada *et al*., 2006). The results of the previous studies on the physico-chemistry of the lake showed that the water quality of Lake Baringo has been deteriorating over time (Oduor *et al*., 2003; Omondi *et al*., 2014; Nyakeya *et al*., 2018). The main concern is on turbidity, which has increased significantly because of high rates of sedimentation from increased soil erosion in the catchment. The turbidity values recorded in 2003 by Oduor *et al*. (2003) ranged between 350 and 900 NTU, which are rather high values than 5 NTU, the standard value for aquatic conditions (Bartram and Balance, 1996; APHA, 2005). Related to the increased turbidity is reduced water transparency of Lake Baringo, which has recorded < 0.1 m, as measured by Secchi disc (Odada *et al*., 2006; Omondi *et al.*, 2014). The overall salinity of the lake water measured as a Na–Ca–HCO_3 composition has not changed greatly since the earliest analyses in 1929 – 1930 (Beadle, 1932; Jenkyn, 1936). From early 1987 to late 2001, the conductivity, salinity, and pH have gradually increased (Barton *et al*., 1987; Aloo, 2002; Oduor *et al*., 2003) with the shrinking volume of the lake. The pH of the lake in 2002 was relatively high varying between 7.77 and 8.91 because of the alkaline hot spring discharge from Kokwa Island, which is located in the lake (Oduor *et al*., 2003; Tarits *et al*., 2006). The high total nitrogen (TN) and total phosphorus (TP) concentrations estimated averagely at 2.8 mg L^{-1} and 1 mg L^{-1} respectively, reflect the lake's hypereutrophic condition (Odada *et al*., 2006; Omondi *et al*., 2016). Temperature measurements in Lake Baringo ranged between 23.7 and 26.3°C (Odada *et al.*, 2006) that could facilitate the growth rates of bloom-forming cyanobacteria such as *Microcystis aeruginosa*. The main factors contributing to the dominance of *Microcystis aeruginosa* in Lake Baringo are temperature and nutrient loading typically TP and TN. The turbid water of Lake Baringo is characterized by a greenish color related to the presence of the cyanobacterium, *M. aeruginosa,* which dominates the lake's phytoplankton community (Ballot *et al*., 2003). As a result of the turbid nature of the lake water, primary production in the open water is very low (Omondi *et al*., 2014). Thus, the phytoplankton community is poor and reported to be limited to the positively buoyant species, comprising *M. aeruginosa*, *M. granulata* and *Anabaena carcinalis (*Kallqvist, 1987). The Lake's high turbidity limits light penetration into the water column, resulting in low biomass production (Odada, *et al.*, 2006). *M. aeruginosa* dominates in Lake Baringo, compared to the other phytoplankton, mainly because it can develop gas vacuoles in its cells, allowing it to regulate its buoyancy (Ballot *et al*. 2003; Odada *et al.*, 2006). The growth rates of bloom-forming cyanobacteria such as *Microcystis aeruginosa* are optimal at 25°C that fall within the temperature range of between 23.7 and 26.3°C reported in Lake Baringo (Odada *et al.*, 2006).

The heavy rains experienced in 2011 in Kenya and the eastern arm of Africa caused the surface area of the lake to increase to 207 km^2 in 2016 (Obando *et al*., 2016). Currently, the concentrations of most physico-chemical variables have been gradually decreasing with the increasing volume of the lake level due to the dilution effect (Nyakeya *et al*., 2020, Walumona *et al*., 2021a). The total dissolved solids (TDS) were reported on an average of 600 mg L^{-1} in most field studies with some values exceeding 800 mg L^{-1} during 1976, 1977, 1984, 1990, and 1996 (Allen and Darling, 1992; Dunkley *et al*., 1993; Darling *et al*., 1996). The TDS values have lately decreased since 2020 to an average of 229 mg L^{-1} (Walumona *et al*., 2021a). A recent study on the spatial and temporal variation in the water quality parameters and trophic status of Lake Baringo showed significant decreases in most of the parameters between 2008 and 2020 indicating the fluctuations of the trophic status of the lake from hypereutrophic to mesotrophic and a great improvement in the water quality of the lake (Walumona *et al*., 2021a).

3.1.6 Sedimentation

Lake Baringo water is highly turbid because of the heavy load of suspended solids carried into the lake mainly by the perennial rivers, and the daily resuspension of the bottom sediments by the wind actions blowing in the late afternoon and early evening (Oduor *et al*., 2003). The sedimentation in Lake Baringo is constituted of fine-grained siliciclastics (Tiercelin, 1981; Renaut *et al*., 2000). The high soil erosion rate in the lake catchment was reported to cover the lake floor by detrital muds and feldspathic silts (Snelder and Bryan, 1995; Oostwoud Wijdenes and Bryan, 2001; Aloo, 2002). The permanent rivers Perkerra and Molo drain a wide portion of the eastern flank of the Tugen Hills and the northern part of the Mau Highlands are the main source of the sediments into the lake (Tarits *et al*., 2006). River Molo is becoming seasonal over the last 3 years (Nyakeya *et al*., 2020, Walumona, Pers. Observation). The lake is also fed by four seasonal rivers (Endau, Lokesen, Mukutan, and Ol Arabel) and streams (Figure 3.3). The two perennial rivers carry a series of basalts, phonolites, and trachytes of Mio-Pliocene age, Pleistocene trachyphonolites, pyroclastic deposits, and siliciclastic fluvial and alluvial sediments into the lake from their upper reaches to their downstream (Snelder and Bryan, 1995). Two seasonal rivers (Mukutan and Ol Arabel) drain the eastern part of the Baringo watershed formed by a succession of basalts and phonolites of Miocene age, several hundred meters thick, which form the Laikipia Fault Escarpment (Renaut *et al*., 2000). The detrital sediment rate is high and highly mixed by the wind actions throughout the year in August during heavy rains resulting in very high turbidity levels (Oduor *et al*., 2003).

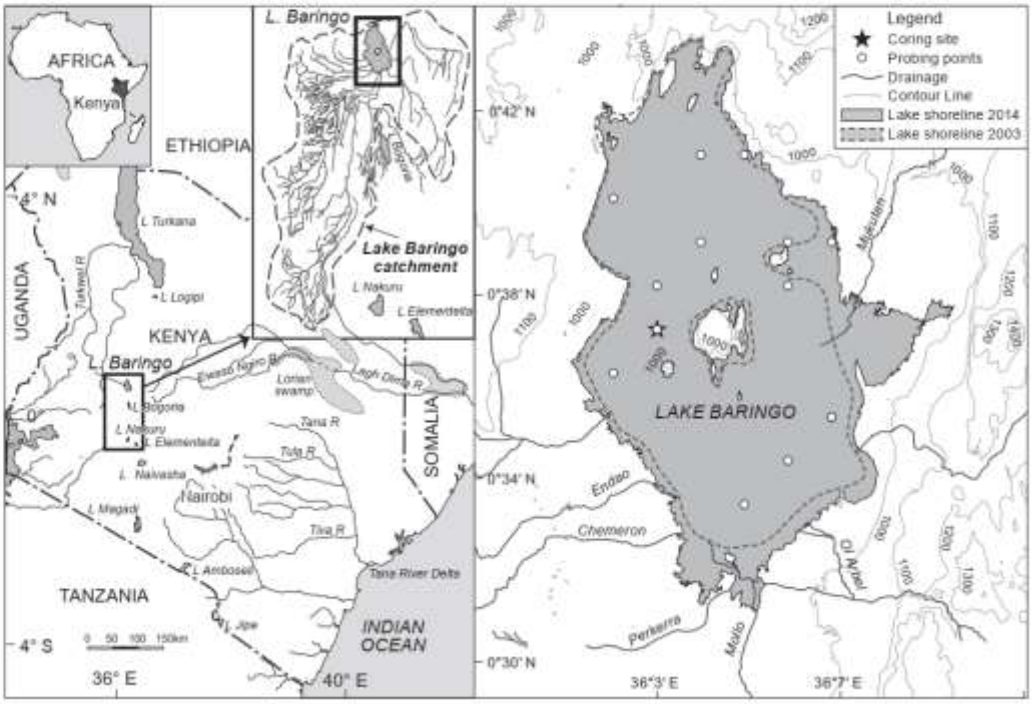

Figure 3.3: Map of Lake Baringo and its drainage basin in the Kenyan Rift Valley with a display of its hydrographic network in relation to those of other rift lakes and lakeshore position in 2003 (stippled line) and during the flooded situation of 2014. Adopted from Okech *et al.* (2019).

The sediment yield of the Lake Baringo basin has been estimated at around 10.38 million MT year^{-1} by extrapolation from erosion studies of the Perkerra catchment (Onyando *et al.,* 2005). For instance, sedimentation is considered to be the main environmental threat to the lake (Odada *et al*., 2006) and reduces both the depth and the surface area of the lake, and destroys the habitats of aquatic animals. The sedimentation of the lake also has increased the pH of the lake's waters in addition to high conductivity and salinity due to the alkaline hot spring discharge from Kokwa Island located in the lake affecting other lake's properties (Hammer, 1986). Land-use changes in the lake catchment (deforestation and agricultural activities) have increased the nutrient levels in the water and stimulated algal production in the lake (Ballot *et al*., 2003). Perkerra river catchment contributes annually to the Lake Baringo sedimentation at 12.8% (Onyando, 2005) of the total annual estimated sediment yield (Aloo, 2002). Since the lake has no surface outflow, this quantity of sediments is trapped in the lake yearly and contributes to the perpetual reduction of the depth of the lake.

3.1.7 Hydrological and climatic conditions

Between 1969 and 1972, records indicated that the average depth of the lake was 8 m (Okech *et al*., 2019). In early 2003, before the onset of the long rains, the average depth was 1.7 m (Onyando, 2005). The average depth increased from 1.7 m to 2.5 m in 2004-2006 with the deepest end of the lake being 3.5 m (Odada *et al*., 2006). This increase in water depth was the result of the prolonged long rains during 2003, especially in the humid upper catchments (Odada *et al*., 2006). Studies by Onyando (2002) demonstrated that the area of the lake was 219 km^2 in 1976, 136 km^2 in 1986, 114 km^2 in 1995, and 108 km^2 in 2001. Based on these trends, the author suggested that the surface area will be reduced by 50% by 2025 if the conditions remained the same (Aloo, 2002; Onyando, 2002; Odada *et al*., 2006).

Although Lake Baringo is located in a semi-arid zone, its catchment covers a range of climatic zones, from semiarid through semi-humid and sub-humid, to a small portion in the humid zone. The mean annual rainfall in the semiarid zone is 450-900 mm and the mean annual potential evaporation amount for this area ranges between 1650-2300 mm (Onyando *et al*., 2005; Odada *et al*., 2006). The risk of crop failure is high and estimated between 25-75% in the semi-arid zone compared to other zones where only between 5-10% is estimated in the semi-humid zone, 1-5% in the sub-humid zone, and < 1% in the humid zone (Kallqvist, 1987). Likewise, the potential growth for plant in these zones ranges from medium to low, high to medium, high, and very high, respectively. These figures indicate that the semi-arid zone, in which Lake Baringo is located, is a fragile environment with low natural life-sustaining properties, thereby requiring urgent conservation attention (Onyando *et al*., 2005).

The rainfall characteristic of the lke's basin is bimodal, intense, and erratic (Ngaira, 2006). The long rains occur from April to August, whereas the short rains fall from October to December with inter-annual variations. Daily rainfall monitoring in the basin dates back to 1903 (Onyando *et al*., 2005). A total of 101 rain gauge stations have been installed and monitored in the catchment by various organizations, including the Kenya Meteorological Department, research organizations, and individuals. However, about 66 stations were operational up to 1926, which gives an approximate gauge density of 97 km^2 $gauge^{-1}$. This number of stations (66) did not meet the World Meteorological Organization's requirement of 17 km^2 $gauge^{-1}$.

Streamflow monitoring started as it was early 1926, with a total of 26 river-gauging stations having been installed at different times in various locations in the rivers flowing into Lake Baringo (Onyando *et al*., 2005; Ngaira, 2006). Most of the above stations are not currently operational because of poor maintenance of the gauges (Odada *et al*., 2006).

The lake is known to has no surface groundwater outflow and is fed by inflows from seasonal (Endao, Lokesen, Makutan, and Ol Arabe) and permenent (Perkerra and Molo) rivers (Figure 2.4), river Molo is becoming seasonal these last three years (Nyakeya *et al*., 200). The lake is believed to have an underground seepage that maintains its freshness by losing approximately 108 m^3 yr^{-1} (Dunkley *et al*., 1993). The streams flowing into Lake Baringo originate from humid and sub-humid hill slopes, where the annual rainfall is more than 1000 mm (Ballot *et al*., 2003; Aura *et al*., 2020).

The hill slopes, which are located in the water recharge areas, have undergone deforestation in the recent past, through land conversion to create more land for agriculture, and through harvesting of forest products for timber, wood fuel, and charcoal.

The forested areas of the catchment have decreased by ≈ 50% since 1976 (Ballot *et al*., 2003). Consequently, groundwater recharge has decreased, with streams drying up more often during the dry seasons, whereas they cause flash floods during the rainy seasons (Ballot *et al*., 2003). Deforestation facilitates the accumulation of greenhouse gases, such as carbon dioxide, in the atmosphere (Ballot *et al*., 2003). These gases can cause global warming and, hence, higher atmospheric temperatures. Increased air temperature due to climate change leads to increased evaporation from the lake which causes a decreased water level in the lake (Odada *et al*., 2006). The decreased water levels have significant impacts, especially on the livelihoods of the communities living downstream and likely the ecology of the lake. This problem is likely to continue in the Lake Baringo basin as long as the population in the upper catchment continues to increase (Odada *et al.*, 2006).

3.1.8 Fisheries and fish biology

Lake Baringo ichthyology has been reported to be poor and composed of five species (Omondi *et al*., 2014). Three species (*Oreochromis niloticus*, *Clarias gariepinus* and *Protopterus aethiopicus*) are of commercial value while, *Labeobarbus intermedius* is rarely appearing in fishermen's catches and *Labeo cylindricus* has almost disappeared in the lake (Aloo, 2002; Mlewa, 2003; Mlewa *et al*., 2006). Traditionally, the lake fishery was mostly based on the endemic *Oreochromis niloticus Baringioensis* (Mlewa *et al*., 2006; Odada *et al.*, 2006) but the fish landings have decreased in recent years due to overfishing and perhaps the cascading effects of environmental changes (Omondi *et al*., 2014). The estimated economic value of the fishery of Lake Baringo for the year 2002 was almost half that of the year 2001, based on experimental fishing during the closure of the fishing activities in the lake. However, in 2001, a fishing moratorium was established, and the estimated economic value of the fishery was US$ 344,560 and derived through interpolation of data (Odada *et al*., 2006). Recently (2020), the fishery landings from Lake Baringo had an economic value estimated at around US$ 1,414,820 which has doubled that of 2015 estimated at US$ 548,596 (Fisheries Bulletin, 2014-2016). The lake fisheries production is locally important, fishing being the main activity, source of animal protein, and income for the riparian community (Aura *et al*., 2020). *L. intermedius* started reappearing in fishermen's catches from 2008 to 2020 (from 0.96% in the 2000s to 3.34% in 2020s) but remains low in the total fishery catches of the lake (Walumona, Pers., Observation).

Recently, some species such as *Labeobarbus intermedius* and *Labeo cylindricus* that had been rarely reported or appear occasionally in catches (Aloo, 2002; Odada *et al.*, 2006; Omondi *et al*., 2014) are currently frequent in the fishers' catches (Walumona, pers. observation). Also, the species composition of commercial catches in Lake Baringo has shifted and is currently dominated by *P. aethiopicus* (61.3 %) followed by *O. niloticus (21.5%)* and *C. gariepinus* (13.9 %). Traditionally, thes fishery was dominated by the native tilapia, *O. niloticus baringoensis* (80.4%) and *C. gariepinus* (9.8 %) up to year 2000s. The dominance of *P. aethiopicus* in the catches from Lake Baringo has also been reported by Mlewa (2003), Mlewa *et al*., (2006), and Aura *et al*. (2020).

The fish community has been very much impacted by both water level changes due to both climate variations and human actions such as upstream river damming and overfishing (Omondi *et al*., 2014; Aura *et al*., 2020; Nyakeya *et al*., 2020). Currently, four fish species (*O. niloticus, C. gariepinus* and *P. aethiopicus and L. intermedius*) are exploited for commercial purposes. All four species are caught by gillnets despite the *P. aethiopicus* that is also caught using hook and longline (Mlewa, 2003, Mlewa *et al*., 2006). Most fishermen use 2.5 mesh sizes of gillnets while the legal fishing regulations recommend the use of only gillnets of 4" and 5" mesh sizes (Aloo, 2002; Fisheries Department, Annual Reports, 2000, 2020). This has led to the landing of small-sized *O. niloticus* in fishermen's catches in addition to the reduction of the fishery production of the lake. The mean size of *O. niloticus* for example decreased to 15 cm in 2001 from 35 cm in 1999, necessitating a fishing moratorium in 2001 (Odada *et al*., 2006). The situation worsened to an extent that the *O. niloticus* in Lake Baringo matured at 13 cm of total length (TL) in 2002 and dropped at 8 cm TL in 2017 with a lower contribution to the lake's total landings suggesting its extinction from Lake Baringo (Britton and Harper, 2008; Tsuma *et al*., 2017; Nyakeya *et al*., 2020). Several reasons have been reported to be the main factors explaining the decline of *O. niloticus* in the fishery of Lake Baringo including environmental stressors, overfishing, climate variability, and insufficiency of food in addition to the presence of the predator and believed invasive (without a conclusive study) species, *Protopterus aethiopicus*, in the lake (Britton *et al*, 2005; Mlewa *et al*., 2006; Britton *et al*, 2009; Omondi *et al*., 2013; Nyakeya *et al.*, 2020).

Labeo cylindricus which migrates upstream of rivers (Molo and Perkerra) to spawn was reported to be close to extinction in the lake (Odada *et al*., 2006). *P. aethiopicus* was intentionally introduced in Lake Baringo in 1975, appeared for the first time in the lake's landing fisheries in 1984, and is currently the dominant fish in the total fish landing at the beaches (De Vos *et al*., 1998; Mlewa and Green, 2004; Mlewa *et al*., 2006; Aura *et al*., 2020; Nyakeya *et al.*, 2020). The fecundity of females in Lake Baringo is low ranging between 77.0 and 125.0-cm with eggs numbering varying from 4179 to 16,528 (Mlewa, 2003; Mlewa and Green, 2004). The population generation time of this fish is relatively long increasing its vulnerability to high exploitation (Mlewa, 2003; Nyakeya *et al.*, 2020). In 1936, studies on the lake's fishery indicated that *Labeobarbus intermedius* (Baringo barb) was the dominant fish species over the entire (Worthington and Ricardo, 1936). Recent investigations on

Labeobarbus intermedius (Baringo barb) suggest that it is functionally extinct from the lake (Odada *et al*., 2006; Omondi *et al*., 2014) and its size has slightly decreased (Odhiambo and Osure, 2021).

Few studies have been conducted so far on the biology and ecology of *Labeo cylindricus* in Lake Baringo where it is almost disappearing in the fishermen's catches.

3.2 Objective 1: Spatio-temporal variations in selected water quality parameters and trophic status of Lake Baringo

3.2.1 Sampling

A total of 27 water samples were collected monthly from 9 stations (C1-S3, Figure 3.1) for each sampling campaign from January 2020-June 2021. Samples were collected in triplicates at each station for the 18 sampling campaigns. The nine representative stations were chosen because of the existence of consistent historical data and they were regularly sampled by Kenya Marine and Fisheries Research Institute (KMFRI) before and after the lake levels dramatically increased over time. The selected nine stations included three sites in the northern part (Stations N1, N2, N3), three stations in the central part (Stations C1, C2, and C3), and three stations in the southern part (Stations S1, S2, S3) (Figure 3.1). Station S2 experienced daily influences of river Molo and partly Perkerra. Station C3 had the influence of rivers Molo and Makutan. C1 was situated on the west adjacent to rocky shores, while N2 lied in the north without any river influence (Table 3.1). In order to study long-term changes, the data collected in this study was complemented by 126 water samples previously collected monthly from five of the nine stations (N2, C1, C2, C3 and S2) by the Kenya Marine and Fisheries Research Institute (KMFRI), Baringo research team from March 2008 to June 2019.

The geographical positions of sampling stations (Table 3.1 and Figure 3.1) were recorded using a hand-held Global positioning System (GPS) navigational unit (Garmin II model). Water samples for nutrient analysis, total suspended solids (TSS), and chlorophyll-*a* were collected monthly directly from the lake surface using clean pre-treated 1-liter polyethylene sample bottles (APHA, 2005). The bottles were labeled, filled and samples were preserved using sulphuric acid (H_2SO_4 50%) then stored in cooler boxes at temperatures of about 4°C, for further laboratory analysis. Monthly average rainfall data for the period 2008-2020 used to assess the influence of rainfall in the lake watershed on the turbidity levels in the lake were obtained from the nearest station within the same basin, Eldama Ravine online station (https://www.worldweatheronline.com/baringo-weather-averages/rift valley/ke.aspx).

Table 3.1: Characteristics of the sampled 9 stations in the three ecological zones in Lake Baringo. Summarized from Several studies (Mlewa, 2003; Odada *et al*., 2006, Mlewa *et al*., 2006; Omondi *et al*., 2014; Nyakeya *et al*., 2020, Walumona *et al*., 2021).

Zone	Station codes	Coordinates (Altitude)	Station names	Shoreline features	Primary Sediment Type (s)	Wind exposure	Bathymetry	Depth range
North	N1	985.8	Loruk	Low influence by small seasonal stream of Katuit	rocky	moderate	Deep area	8.2-13.9
	N2	986.1	Lokoros	No river disturbance	rocky	Extreme	Deepest area	11.3-15.4
	N3	985.4	Komolion	Influenced by the seasonal river of Mukutani	rocky	moderate	Deep area	10.4-15.0
Central	C1	982.4	Ngenyin	East-west transects and west adjacent to rocky shores with ngenyin seasonal river	rocky	moderate	Moderate depth	10.6-14.0
	C2	987.0	Samatian	At the center of the lake	muddy	Extreme	Gradual slop	10.0-14.5
	C3	988.3	Ruko Conservancy	Influenced by Hot springs and to some extent rivers Molo and Mukutani	muddy	moderate	Moderate depth	7.0-13.4
South	S1	983.9	Salabani	Influenced by Rivers Endau & perkerra with a few settlements	muddy	moderate	shallow area with macrophytes	6.3-10.2
	S2	983.	Molo river mouth	Macrophytes and daily influences of 2 rivers Molo and partly Perkerra	muddy	Extreme	shallowest area with macrophytes	4.4-7.5
	S3	990.8	Kiserian	Influenced by Arabel river plus few settlements	muddy	moderate	shallow area with macrophytes	6.2-11.9

3.2.2 Analytical procedures

Measurements of pH, total dissolved solids (TDS), dissolved oxygen (DO), temperature (T), and electrical conductivity (EC) were carried out *in situ* with the Professional Plus multi-parameter instrument (YSI 550) calibrated with standard solutions (SMEWW, 1998). *In situ* measurement of turbidity (TUR) was performed using a calibrated portable turbidimeter probe (HACH 21000Q). Before measurements, the turbidimeter was calibrated using standard chemical solutions made of stabilized formazin with turbidity values of 20, 100, and 800 NTU (HACH, 2009). Samples with high turbidity exceeding 800 NTU were diluted with distilled water to make them fit into the turbidimeter reading capacity limits (HACH, 2009). Then, the turbidity of diluted samples was estimated by multiplying the actual reading by the number of dilutions applied on the sample. Water transparency (m) was measured by a standard Secchi disk of 20 cm diameter. The effective Secchi depth was calculated as the average of the depth at disappearance and that of reappearance of the disk. The obtained Secchi disk values were used to calculate the euphotic zone (Zeu) of the lake at each sampling station. The euphotic zone (Zeu), defined as a depth at which photosynthetically available radiation (PAR) is 1% of its surface value, was estimated by using the formula:
Zeu = 4.6/k (Bartram and Balance, 1996), and derived from estimates of the vertical light extinction coefficient from Secchi disk transparency using a coefficient k (k = 1.5/Secchi disk depth in meters) at each sampling station.

Ammonium, orthophosphate, total phosphorus, nitrite, nitrate, total nitrogen, and ractive soluble were determined usind various methods following the procedure of

Bartram and Balance (1996), and APHA (2005). Soluble Reactive silica (SRSi) was determined using the molybdate complex method while, Soluble Reactive Phosphorus (SRP) was determined using the molybdenum blue method (APHA, 2005). Ammonium (NH_4^+) was determined by the dichloroisocyanurate – salicylate method, nitrate (NO_3^-) by the cadmium reduction method, and nitrite (NO_2^-) from the azo – dye complex formation (APHA, 2005; Rodier *et al*., 2009). Beer-Lambert law ($A = \varepsilon l C$) (Bartram and Ballance, 1996) and APHA (2005) (A = absorbance, l = cuvette width, ε = extinction coefficient) was used to convert the obtained absorbance of each nutrient to its corresponding contracentration.

Total phosphorus and total nitrogen were analyzed by the ascorbic acid reduction method and diazotization method, respectively, using unfiltered water (APHA, 2005). Total alkalinity was estimated by the volumetric method, using sulphuric acid (H_2SO_4), phenolphthalein and methyl orange indicator (APHA, 2005). Fluoride ion (F^-) concentrations were analyzed using titrimetric methods based on titration of a sample with silver nitrate (Bartram and Balance, 1996). Total suspended solids (TSS) were determined by filtration of a known volume of the lake water through GF/F filter which was firstly dried and thereafter pre-weighed and then oven-dried after filtration and final weights were taken to determine the difference as the TSS weight (g) per unit volume of the sample (Rodier *et al*., 2009). Total dissolved solids (TDS) were determined using gravimetric analysis based on filtration and gravimetric methods by the temperature-controlled oven.

The water total hardness (TH) was determined by a complexometric method using EDTA solution. The analytical methods and procedures used are described in detail by APHA (2005). In the laboratory, water samples for above nutrient analysis were filtered using GF/F filter papers (47 mm in diameter and 0.7 μm pore size) within 24 hours from collection. Water samples for water quality analyses were shipped refrigerated from KMFRI Baringo Station to KMFRI Kisumu whenever water quality analyses were not possible in Baringo.

3.2.3 Determination of Chlorophyll-a (Chla)

The water samples for chlorophyll-a (Chl-a) analysis were collected at each station using a small 5-liter polyethylene canister, previously rinsed with distilled water. Before Chl-a analysis, water samples were filtered using Macherey-Nagel GF/F filters with a porosity of 0.7 μm and 47 mm in diameter. The labeled vials were placed in a portable freezer at 4°C and later stored in a fridge at -20°C for further chlorophyll-a analysis.

The chlorophyll-*a* concentration was determined through three steps (APHA, 2005) as follows:

a) the extract in acetone was sonicated once, kept at 4°C protected from light for 12 hours,

b) the extract was sonicated a second time for a complete transformation, and

c) the algal concentration was finally calculated according to the Lorenzen equation (APHA, 2005) from the absorbance values of the chlorophyll-a extracts before and after acidification with 0.1N HCl using the following formula:

$$\text{Chl. a } (\mu g) = 11.9 * [2.43\ (D_b - D_a)] * \left(\frac{V}{L}\right) \qquad (1)$$

Where, Db = optical density before acidification; Da = optical density after acidification; V: a volume of solvent (acetone) (in mL); and L = thickness of the spectrophotometer cuvette. The absorbance was transformed into concentrations using the Beer-Lambert method (Brtram and Balance, 1996).

3.2.4 Planktonic population sampling

3.2.4.1 Water sample collection and conservation

Planktons are a heterogeneous group of organisms that include both phytoplankton and zooplankton. Triplicate samples for both phytoplankton and zooplankton were collected using a conical net with a mesh size of 60 µm and a diameter of 30 cm at the opening. The net was towed horizontally to a 3 m distance from the marked rope. The net was towed and washed after each collection to rinse off any plankton (zoo and phyto) that could remain in the net body. After collection, the material retained in the net was kept in 400 mL plastic bottles and fixed in Lugol's solution before analysis. At the lower end of the plankton net, a graduated glass bottle is fitted to retain sedimented planktonic organisms. The final volume (125 mL) of the filtered sample was transferred to another plastic bottle of volume 125 mL which was labeled indicating the time, date, and name of the sampling station.

The plankton (both zoo and phyto) samples kept in 125 mL plastic bottles were preserved by the addition of 3-5 drops of Lugol's solution. After adding the Lugol's solution, the preserved samples were kept for 24 hours undisturbed allowing the sedimentation of suspended plankton in the water before taking the samples for phytoplankton and zooplankton identification. After 24 hours, a dropper was used to carefully remove the supernatant without disturbing the sediments and 50 mL of concentrated sample was taken for plankton (phyto and zoo) analysis. The phytoplankton and zooplankton densities in the samples were quantified using the formulae in Rahimibashar *et al*. (2009) and Niyoyitungiye (2019).

3.2.4.2 Quantitative analysis of planktons

Sedgwick-Rafter cell and Lackey's drop methods (Onyema, 2008; Niyoyitungiye, 2019) were used for both qualitative and quantitative planktonic analyses (Appendex 1). Sedgwick-Rafter cell method is used when the density of plankton and filamentous microalgae are less abundant in the sample, while Lackey's drop method is commonly used for samples with high plankton densities (Kamaruzzaman *et al.,* 2010; Niyoyitungiye, 2019). The quantitative analysis of plankton estimated the numbers of individuals observed under light microscope for each taxonomic group and the number of organisms was expressed in total organisms per liter using the Sedgwick-Rafter's formulas for zooplankton (Niyoyitungiye, 2019). Among phytoplankton, many are multi-celled filamentous, others are colonized, while some are solitary cells. Thus they are more conveniently expressed as units/Liter in abundance (Kamaruzzaman *et al.,* 2010).

Plankton taxa were counted in subsamples of 1-3 mL, depending on their density, using a plastic pipette and a gridded counting chamber under an optical microscope (x 25). During analysis, the effect of surface tension on the specimens was minimized by adding some drops of liquid detergent whereas drops of Lugol's solution were used to improve the visibility.

Both zooplankton and phytoplankton were identified using relevant taxonomic literature (Culver *et al.*, 1985; online source in Appendex 1). Korovchinsky (1992), and Smirnov, and Timms (1983) were used in Cladocera identification, while Koste and Shiel (1986), and Segers (1995; 2007) were used to identify the Rotifera community. The zooplankton was identified up to a taxonomic precision of order-level. Phytoplankton and zooplankton biomass (g m^{-2}) estimates followed Jones (1979), Culver *et al.* (1985), and Azevedo *et al*. (2012) methods.

3.2.5 Detritus determination

The primary production (PP) estimates and euphotic zone (Zeu) were used to estimate the detritus biomass (g C m^{-2} yr^{-1}) for 1999, 2010, and 2020 based on Christensen and Pauly's equation (Christensen and Pauly, 1993) as follows:

$$\mathrm{Log}(D) = [0.954\times\log(\mathrm{PP})] + [0.8631\times\log(Z_{eu})] - 2.41 \quad (2)$$

Where: D = stock of detritus (g C m^{-2}); PP = primary production (g C m^{-2} yr^{-1}); and Zeu = euphotic zone (m).

The euphotic zone (Zeu), defined as the depth at which photosynthetically available radiation (PAR) is 1% of its surface value, was estimated as Zeu = 4.6/*k* (Bartram & Balance, 1996) and derived from estimates of the vertical light extinction coefficient from the Secchi disk transparency using a coefficient *k* (*k* = 1.5/Secchi disk depth in metres) at each sampling site.

Then, the calculated detrital biomass used for the three models was 0.35 g C m^{-2} $year^{-1}$ for 1999 trophic model, 0.32 g C m^{-2} yr^{-1} for 2010 trophic model and 152.70 g C m^{-2} yr^{-1} for 2020 trophic model, respectively.

The data on phytoplankton, zooplankton, and detritus were used in the Ecopath with Ecosim model (see section 4.4).

3.2.6 Calculation of Water Quality Index (WQI) of Lake Baringo

The water quality index (WQI) is a rating that reflects the composite influence of different water quality parameters (Sahu and Sikdar, 2008; Ramakrishnaiah *et al*., 2009). Firstly, each of the chemical parameters (e.g. pH, TDS, turbidity, total alkalinity, hardness, fluoride, nitrate, nitrite, silica, phosphate) is assigned different weights (w_i) on a scale of 1 (least effect on water quality) to 5 (highest effect on water quality) based on its perceived effects on human health and according to its relative importance in the drinking water or groundwater quality (Brown *et al*., 1972; Sener *et al*., 2017). The highest weight of 5 is assigned to parameters that have critical health effects and whose presence above the critical concentration limits could hinder the usability of water for domestic and drinking purposes (Bhateria and Jain, 2016; Sener *et al*., 2017). In this study, nutrients and fluoride were assigned the highest weight (5) because of their health influence (WHO, 2008) and importance in water quality assessment while, a minimum weight of 1 was assigned to total alkalinity and electrical conductivity parameters due to their least importance in water quality assessment (Brown *et al.*, 1972; Katyal, 2011).

The relative weight (RW_i) which is the contribution of each parameter to the WQI was then computed from the following equation (Sener *et al.*, 2017):

$$RW_i = \frac{w_i}{\sum_i^n x W_i} \qquad (3)$$

Where, w_i is the assigned weight of each parameter (Table 3.2) and n is the number of parameters. Then, a quality rating (Q_i) for each parameter except pH and DO was assigned by dividing its concentration (C_i) in each water sample by its limits values/standards given by the WHO (2008, 2011) and the result multiplied by 100:

$$Q_i = \frac{C_i}{S_i} \times 100 \quad (4)$$

Where:

Q_i = the quality rating,

C_i = the concentration of the chemical parameter in each water sample in mg/L, and

S_i = is the drinking water standard for the chemical parameter in mg/L according to the guidelines of WHO (2008).

Table 3.2. Variables used in the water quality index calculation, scores of normalization (*Ci*) and relative weights (*Pi*).

Variable	Units	Relative weight (*Wi*)	Normalization factor (*Ci*)										
			100	90	80	70	60	50	40	30	20	10	0
Temp.	°C	3	21/16	22/15	24/14	26/12	28/10	30/5	32/0	36/−2	40/−4	45/−6	> 45/< -6
pH	Unit pH	1	7	7–8	7–8.5	7–9	6.5–7	6–9.5	5–10	4–11	3–12	2–13	1–14
EC	μS/cm	1	< 750	< 1000	< 1250	< 1500	< 2000	< 2500	< 3000	< 5000	< 8000	≤ 12000	> 12000
DO	mg/L	4	≥ 7.5	> 7	> 6.5	> 6	> 5	> 4	> 3.5	> 3	> 2	≥ 1	< 1
TDS	mg/L	2	< 100	< 500	< 750	< 1000	< 1500	< 2000	< 3000	< 5000	< 10000	≤ 20000	> 20000
Turb.	mg/L	4	< 5	< 10	< 15	< 20	< 25	< 30	< 40	< 60	< 80	≤ 100	> 100
SiO_4^{4-}	mg/L	1	< 25	< 50	< 75	< 100	< 150	< 250	< 400	< 600	< 1000	≤ 1500	> 1500
PO_4^{3-}	mg/L	1	< 0.025	< 0.05	< 0.1	< 0.2	< 0.3	< 0.5	< 0.75	< 1	< 1.5	≤ 2	> 2
NH_4^+	mg/L	3	< 0.01	< 0.05	< 0.1	< 0.2	< 0.3	< 0.4	< 0.5	< 0.75	< 1	≤ 1.25	> 1.25
NO_2^-	mg/L	2	< 0.005	< 0.01	< 0.03	< 0.05	< 0.1	< 0.15	< 0.2	< 0.25	< 0.5	≤ 1	> 1

EC: Electrical conductivity; Turb.: turbidity and Tem.: temperature

The quality rating for pH or DO (QpH, DO) was calculated following Alobaidy *et al.* (2010) as:

$$Q_{p^h}, DO = \frac{[C_i - V_i]}{[S_i - V_i]} \times 100 \quad (5)$$

Where, V_i = the ideal value which is considered as 7.0 for pH and 14.6 for DO (WHO, 2011).

Equations (4) and (5) ensure that $Q_i = 0$ when a pollutant is totally absent in the water and $Q_i = 100$ when the value of this parameter is just equal to its permissible value (Bhateria and Jain, 2016). Hence, the higher the value of Q_i, the more polluted is the water.

Consequently, to calculate WQI, firstly, sub-index (SI_i) value is determined for each water quality parameter and then used to derive WQI with the following equations (Bhateria and Jain, 2016; Kumar *et al*., 2018):

$$SI_i = RW_i \ x \ Qi \tag{6}$$

$$WQI = \sum_1^n x \ Sl_i \tag{7}$$

Where, SI_i is the sub-index of ith parameter; Q_i is the quality rating based on the concentration of ith parameter. The computed WQI values were classified into five categories ranging from 1 (excellent) to 5 (unsuitable for drinking) as in Table 3.3 following the equation proposed by Bhateria and Jain (2016). Thus, the highest WQI reflects the poorest water quality of the lake in space and time.

Table 3.3: Water quality classification based on WQI values (Bhateria and Jain, 2016)

N°	WQI Values	Water quality
1	< 50	Excellent water
2	50 – 100	Good water
3	100 – 200	Poor water
4	200 – 300	Very poor water
5	> 300	Unsuitable for drinking

The effective weight (Ewi) of each water quality parameter corresponds to the individual parameter weightage compared to the overall weight on the water quality. Ewi were calculated using the derived WQI values. The effective weight for each parameter was derived by dividing its sub-index value (SIi) by the overall Water Quality Index value and the result multiplied by 100 as (Sener *et al*., 2017):

$$EW_i = \frac{SI_i}{WQI} \times 100$$

(8)

Where Ew_i is the effective weight of ith parameter; SI_i is the sub-index value of ith parameter (Equation 6) and WQI is the overall Water Quality Index computed by Equation (7). The relative weights were compared with effective weights to reflect the significance of each parameter compared to other parameters used in WQI calculations.

3.2.7 Calculation of Organic Pollution Index (OPI)

The organic pollution index (OPI) of Lake Baringo, a measure of organic load, was calculated for each sample collected at all the five stations following the procedure described by Bahroun and Bousnoubra (2011). The OPI derivation is based on the calculation of the average values of four parameters which are: biological oxygen demand (BOD), ammonium, nitrite, and phosphate (Bartram and Balance, 1996). The concentrations of the four parameters are compared with the standard limits to determine the parameter class numbers (Leclercq and Maquet, 1987; Rodier *et al*., 2009) (Table 3. 4 and Table 3.5). For this study, the OPI class for the samples was assessed using three nutrient variables (ammonium, nitrite, and phosphate) of the recommended parameters. Biochemical oxygen demand *(*BOD) was not estimated during data collection and analysis. The OPI was then calculated as the average of the class numbers (Table 3.4) of the three parameters used in this study (Bahroun and Bousnoubra, 2011). Thus, the lower the value of OPI, the more organically polluted is the water body based on categories shown in Table 3.5.

Table 3.4: Parameter classes and limits for OPI index calculation. Adopted from Leclercq and Maquet (1987)

Parameters Classes	ammonium mg –N/L	nitrite µg-N/L	phosphate µg-P/L
5	< 0.1	5	15
4	0,1– 0.9	6 – 10	16 – 75
3	– 2.4	11 – 50	76 – 250
2	2.5 – 6.0	51 – 150	251 – 900
1	> 6	> 150	> 900

Table 3.5: Categories of water pollution based on the OPI index and watercolors. Adopted from Leclercq and Maquet (1987)

Category of pollution	OPI	Colors allocated to the index
Null pollution	5.0 – 4.6	Blue
Weak pollution	4.5 – 4.0	Green
Moderated pollution	3.9 – 3.0	Yellow
Strong pollution	2.9 – 2.0	Orange
Very strong pollution	1.9 – 1.0	Red

3.2.8 Lake Baringo trophic status estimation

The Trophic Status of Lake Baringo was evaluated using information collected on the concentration of the nutrient (total phosphorus), chlorophyll-a as an indicator of phytoplankton biomass, and water transparency (dependent on both algal biomass and sediment resuspension, expressed as Secchi depth) ***(***Istvánovics, 2010***).*** The nutrient availability was assessed using concentrations of readily bioavailable inorganic nutrients: TN:TP ratio and SRP:DIN ratio comprising (DIN: nitrate, nitrite and ammonium (OECD, 1982; Reynolds, 1999). N limitation was considered probable when molar TN:TP < 10 and P limitation when TN:TP > 20 (Stephen *et al*., 2020). The intermediate ratios indicate potential co-limitation between TN and TP (Stephen *et al*., 2020). Additionally, the Carlson Trophic State Index (CTSI) (Carlson and Simpson, 1996) was also used to evaluate and classify the trophic status of the lake (Table 3.6). The accepted standard limits used to evaluate the lake's trophic status were adopted from the Organization for Economic Cooperation and Development (OECD) for the individual parameters (total phosphorous, chlorophyll-a, and Secchi depth), and following Carlson and Simpson (1996).

To quantitatively assess the trophic status of a lake ecosystem and/or a reservoir, a Trophic State Index (TSI) based on the values of individual parameters and with a scale of 0 – 100 was used (Carlson and Simpson, 1996). The TSI value obtained facilitates qualitative descriptions of the lake trophic status (Table 3.6).

The TSI is split into five groups: 0–20, 20–40, 40–60, 60–80, and 80–100 corresponding to five lake trophic states, hyper oligotrophic, oligotrophic, mesotrophic, eutrophic, and hypereutrophic, respectively (Likens *et al.,* 1977; Al-Haidarey *et al*., 2016).

The Carlson's Trophic State Index (CTSI) (Carlson and Simpson, 1996) was calculated based on the average TSI of the individual parameter values using the following formulae:

$$TSI for chlorophyll - a\ TSI(chl - a) = 9.89 * Inn[chl - a(\frac{\mu g}{L})] + 30.6 \qquad (9)$$

$$TSI for\ Secchi\ depth\ TSI(SD) = 60 - 14.41\%\ (SD)(m) \qquad (10)$$

$$TSI for\ Total\ phosphorus\ TSI(TP) = (14.42 * TP(\frac{\mu g}{l})) + 4.15 \qquad (11)$$

The Carlson's Trophic State Index (CTSI) was then obtained by calculating the average of the equations 9, 10, and 11 (Carlson and Simpson 1996):

$$(CTSI) = \frac{[TSI(TP) + TSI(Chl-a) + TSI(SD)]}{3} \qquad (12)$$

Based on the derived CTSI, the trophic state of Lake Baringo was determined according to Table 3.6 (Carlson and Simpson, 1996).

Table 3.6: Parameter trophic state indices and Carlson's Trophic State Index values for classification of lakes (Carlson and Simpson, 1996). SD = Secchi depth; TP = Total Phosphorus; Chl-a = Chlorophyll-a values; CTSI = Carlson Trophic State Index.

SD (m)	TP (µg/L)	Chl-a(µg/L)	CTSI	Lake trophic state	Attributes
> 8	< 6	< 0.95	< 30	Oligotrophic	With clear water, oxygen throughout the year in the hypolimnion
8 – 4	6 – 12	0.95 – 2.6	30 – 40	Oligotrophic	Oligotrophy, but some shallower lakes will become anoxic during the dry season
3.9 – 2	12.1 – 24	2.6 – 7.2	40 – 50	Mesotrophic	Water moderately clear, but increasing of anoxia during the dry season
1.9 – 1	24.1 – 48	7.2 – 25	50 – 70	Eutrophic	Decreased transparency, warm-water fisheries only
0.9 – 0.5	48.1 – 98	25 – 55	70 – 80	Eutrophic	With possibility of Heavy algal blooms during the dry season with the tendency of becoming hypereutrophic
< 0.5	> 98	> 55	> 80	Eutrophic (Hypereutrophic)	Reduction in macrophyte species, algal scum and loses in fish stocks in dry season

3.3 Objective 2: Lake water balance modelling and its sensitivity to hydro-meteorological variations

3.3.1 Lake Water Balance Model (LWBM)

The lake water balance is defined as the exchange of the water mass between input and output over the lake and its catchment area (Zhang *et al*., 2015). The lake water balance is also considered as the magnitude and timing of each of the flows entering and leaving the lake (Dessie *et al*., 2015). A water balance model for the lake's system was used to evaluate the exchange of water between the different components of Lake Baringo.

The lake water balance modeling was estimated by including inflow and outflow components represented by Eq. (13) and (14). Two models were run, the first one for the period from 1970-1995 based on existing data (Ngaira, 2006; Aura *et al*., 2020) and the second one for the period from 2008 to June 2021 based on the short-term data collected by this study from January 2020 to June 2021. The two models incorporated all component values estimated following the described steps and procedures (Mbanguka *et al*., 2016; Zhang *et al*., 2002; 2015).

$$\frac{\boldsymbol{dL}}{\boldsymbol{dt}} = RL\ (t) - EvapL + Qbala + \ \varepsilon\ (t) \qquad (13)$$

and

$$A_L\ x\ [(P_{(t)} + Qr_{\ (t)}) - (E_{(t)} + Qout_{(t)})] = Qbala_{(t)} \qquad (14)$$

Where, $Qbala$ (mm yr^{-1}) is net runoff, RL is the possible runoff volume to the lake (mm yr^{-1}), A_L is the lake surface area (m^2), $P(t)$ precipitation (mm yr^{-1}) at time t, $Qr_{(t)}$ surface runoff entering into the lake at time t, $EvapL(t)$ *or* (E_t) (mm yr^{-1}) water lost from the lake surface due to evaporation at time t, $P(t)$ is the rainfall (mm yr^{-1}) on the lake surface at time t (yr), L is the water level (m) in the lake and ε represents uncertainties in the water balance arising from errors in the data and/or other terms such as minor abstraction or inflow from ungauged catchments.

In this study, the model was done at a yearly time step (t). $Qout_{(t)}$ (Equation 14) is the discharge of the outflow of the lake due to water abstractions from the inflows and the underground seepage (S) and was estimated as $Q_{out}(t) = A_{(t)} + S_{(t)}$.

S (t) is the loss due to the underground seepage estimated at 108 m^3yr^{-1} (Dunkley *et al*., 1993) and $A_{(t)}$ losses due to river water abstractions (mm yr^{-1}). All the lake water balance components have been converted in the same units for comparisons purposes. The stored water in the lake (storage rate; S) per time unit (S = total inflows – total outflows) is the difference between the total inflows in the lake and total outflows from the lake over time.

3.3.2 Determination of parameters for water balance model

3.3.2.1 River sampling for velocity and discharge

Four rivers (Perkerra, Molo, Endao, and Lokesen, Figure 3.1) in the Lake Baringo basin were sampled for the discharge measurement into Lake Baringo from January 2020 to June 2021 on a monthly schedule. These rivers were selected among the seven rivers flowing into the lake (Aloo, 2002; Odada *et al*., 2006; Nyakeya *et al*., 2020) according to their perennity, size, geographical location, accessibility, and prevailing land use. Additional long-term data on river discharge were obtained on two permanent rivers (Perkerra and Molo) from 1970-1995 based on the literature (GOK, 1989, 2002, 2007; Akivaga, 2010). The selected four rivers drain ~ 1765 km^2

in the southern region, representing ~75% of the river-active area catchment of around 2294.5 km^2 (the total catchment of the lake is 6820 km^2, Table 3.7). Other geomorphological, physical, and chemical properties of the lake are detailed in Table 3.6.

Table 3.7: Morphometric and physico-chemical characteristics of Lake Baringo (Kenya), data from various sources

Morphometric characteristics		Physico-chemical characteristics	Range
Maximum depth (m)	~ 20 [f]	Surface temperature (°C)	23.2-30.2°C [f]
Mean depth (m)	10.8 [f]	Conductivity (µS/cm)	303.3-846.2 [f]
Lake surface area (AL) (km^2)	250 [g]	Turbidity (NTU)	5.7-481,9[f]
Lake basin area (AB) (km^2)	6820 [c]	pH	7.1-9.8 [f]
Northern and central areas without rivers (km^2) (AWR) (AC-AR)	4525.5 [f]	Na+ (meq L^{-1}) (mean)	15.5 [h]
River active area (Southern part)(AR) (km^2)	2294.5 [f]	K+ (meq L^{-1}) (mean)	0.6 [h]
Catchment area without lake (AC) (= AB-AL) (km^2)	6570 [f]	Ca++ (meq L^{-1}) (mean)	0.6 [h]
Lake volume (VL) (km^3)	930 [b]	Mg++ (meq L^{-1}) (mean)	0.2 [h]
Subaquatic seepage (m^3yr^{-1})	108 [d]	Cl^- (mg L^{-1}) (mean)	3.2 [h]
Water residence time (Rt) (months) (VL/Tot inflows)	[f]	TDS (mg L^{-1})	90.1-471.1 [f]
Major water use	Navigation, fishing, drinking [b]	Surface DO (mg L^{-1})	4.5-8.7 [f]
Altitude (m)	97 [e]	Chlorophyll a (µg L^{-1})	2.1-39.6[f]
Latitude	0°30’N and 0°45’ N [e]	Alkalinity (mg L^{-1} CaCO3)	79.5-312.0[f]
Longitude	36° 00’ E and 36° 10’ E [e]	Secchi depth (cm)	83.8-182.5[f]
AL/AC	0.03[f]	TN (µg L^{-1})	1.2-733.3[f]
Runoff coefficient (k)	1.48 [f]	TP (µg L^{-1})	8.3-292.6[f]

(a) : Omondi *et al*., 2014; (b): Obando *et al*., 2016; © : Odada *et al*., 2006; (d) Dunkley *et al*., 1993; (e) Kallqvist, 1987; (f) this study; g) Nyakeya *et al*., 2020, h) Oduor *et al*., 2003. River active area in this study takes into account the two perennial rivers Molo and Perkerra and two seasonal rivers Endao and Lokesen areas in the lake watershed.

The geographical coordinates of the river sampling locations were recorded using a Garmin Olathe 72 hand-held GPS (Garmin, USA). Three rivers (Molo, Endau, Lokesen) were sampled at a distance of ~100 m from the mouth, while river Perkerra was sampled far away from the mouth at 5 km because of difficulties with near-shore accessibility. River discharge was measured by determining the velocity of a floater and the total cross-sectional area of the river following the Float Method Procedure (Harrelson *et al.* 1994; Bartram and Balance, 1996). A correction factor of 0.85 was used to adjust the calculated velocity of the river surface (Harrelson *et al.* 1994; Rodier *et al.*, 2009).

The river width was measured by a 50 m-long tape measure and then divided into sub-sections for which the depths were determined according to the procedure in Bartram and Balance (1996). Six to seven vertical sub-sections were selected along the cross-section of the river depending on the width and shape at the site of each river. A 50 m-long tape measure was used to also measure the length (L) (in meters) of floatation of a float in a given period (t) (in seconds) measured using a chronometer. A stick meter allowed the determination of the width of each vertical segment of the cross-section of sites (Figure 3.2).

The discharge was computed at each site cross-section based on the assumption that the average velocity measured on a vertical line is valid for a rectangle that extends half of the distance to the verticals on each side of it, as well as throughout the depth at the vertical plane (Harrelson *et al*. 1994; Bartram and Balance, 1996). In Figure 3.2, the mean velocity (vn) ($v_n = b_n/t_n$) applies to each rectangle bounded by the given colored line. The area (a_n) of each rectangle is given in the function of the width (b_n) and the depth (d_n) at each sub-section by (Harrelson *et al*. 1994):

$$a_n = d_n * \frac{[(bn+1)-(bn-1)]}{2}; \qquad (15)$$

and the discharge (Qn) through it was estimated using the following equation:

$$Q_n = a_n * v_n \qquad (16)$$

Therefore, the total discharge (Qt) across the entire cross-section of the river was given by summation:

$$Qt = Q_1 + Q_2 + Q_3 + \ldots Q_{(n-1)} + Q_n \qquad (17)$$

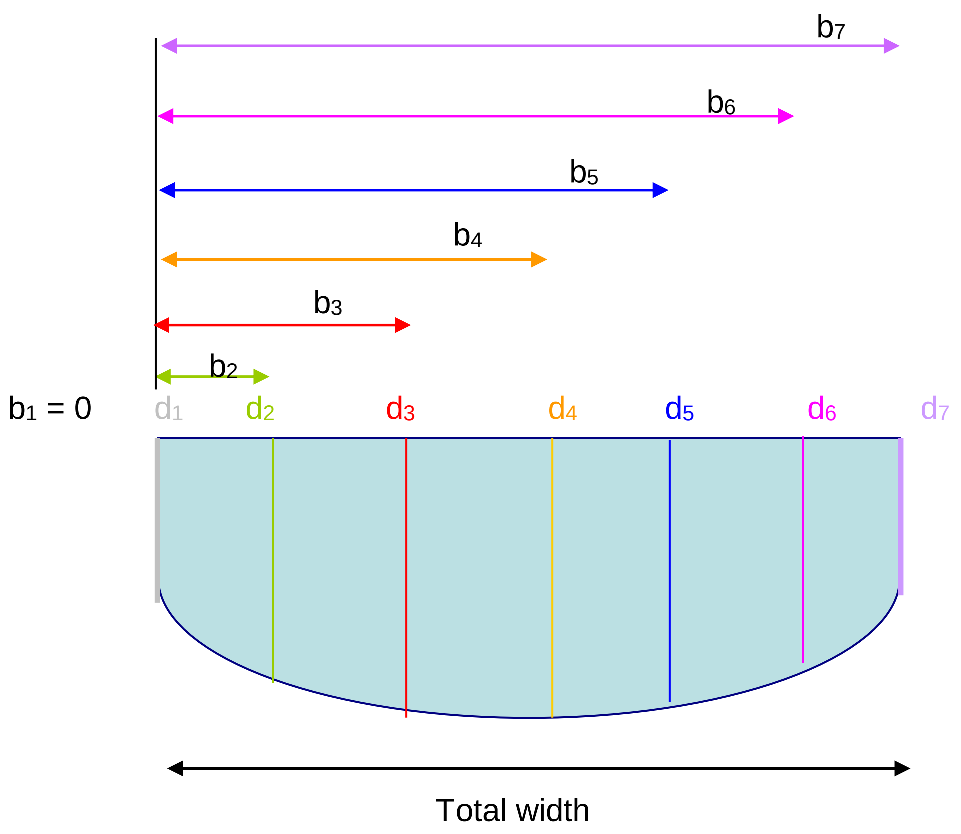

Figure 3.2: Cross-section of a stream divided into vertical sections for measurement of discharge (Bartram and Balance, 1996)

3.3.2.2 Outflow and Water Extraction volumes for Irrigation Supply

Lake Baringo has no known surface outflow; however, water is being extracted from the lake's watershed for irrigation purposes. The volumetric method was adopted to estimate Lake Baringo water extraction for dams in the lake watershed for irrigation such as the Perkerra irrigation scheme by the urban population. The average pump discharge at two inflows (Perkerra and Molo rivers, Figure 3.1) was measured based on daily observations and recorded pump operational hours. Water extraction quantities (I_i) from the ith inlet for the entire crop season (for irrigation) were estimated from the following empirical formula (Törnqvist and Jarsjö, 2012; Machiwal *et al.*, 2016):

$$I_i = \sum_{j=1}^{ni} qij \text{ x } dij \quad (18)$$

Where q_{ij} pump discharge at ith inlet on jth day (m^3 h^{-1}), dij operational pumping hours at ith outlet on jth day (h), n_i is the number of days when pump at ith inlet was operating during a crop season and i inlet number.

3.3.2.3 Evaporation estimation over Lake Baringo

The evaporation losses from lakes have been determined with various methods estimating the evaporation losses from an open or free water surface (Singh and Kar, 1996; Tanny *et al*., 2008). The Penman method (Penman, 1948), Thornthwaite method (Mather, 1978; 1981), and the energy balance method (Rosenberry *et al*., 2007) have been widely used in the estimation of evaporation in studies of East African lakes (Yin *et al*., 1998; Kebede *et al*., 2006; Mbanguka *et al.*, 2016). Due to data availability constraints, the semi-arid location of Lake Baringo, and the complexity of methods in terms of inputs, the Thornthwaite Method, a very simple and data-poor approach based only on air temperature data, was used at a monthly time-step for annual evaporation estimates of Lake Baringo for water balance modeling and sensitivity analysis. Moreover, despite the perceived underestimation of the evaporation outcome by this method (Mbanguka *et al.*, 2016), it is well adapted and largely used in semi-arid areas (Rosenberry *et al*., 2007).

The climatological data (air temperature) measured in the lake watershed were collected from Eldama Ravine online station (ELDR, https://www.worldweatheronline.com/baringo-weather-averages/rift-valley/ke.aspx) and air temperature measured at around 2 meters above the lake surface were as well collected from various literature for the period from 2008 to 2019 (Omondi *et al*., 2014; KMFRI, unpublished reports, 2020) and complemented with additional data collected in this study from 2020-2021. The data sources were averaged to get the representative estimates of air temperature on the Lake Baringo watershed that were used for the estimation of evaporation over the lake. The Thornthwaite Method was then used to estimate the lake potential evapotranspiration (PET) and active evapotranspiration (*ETP*). The monthly evaporation rate (E) from the lake was determined as proposed by Mather (1978; 1981) using Thornthwaite's equation and variables below:

$$\mathrm{PET} = [1.6\ (10\ \frac{\mathrm{T}}{\mathrm{I}})^{a}\ (\frac{10}{\mathrm{d}})] \quad (19)$$

Where, I is the annual heat index, T is the air temperature in °C, d is the number of days in a month and the parameter "a" is a function of I (Mather, 1978; 1981) as.

$$\mathrm{I} = \sum_{\mathrm{i}=1}^{12} i \quad (20)$$

and

$$i = [\frac{\mathrm{T}}{5}]^{1.514} \quad (21)$$

Where, *i* is the heat index per month.

$$a = 6.75{*}10^{-7}\mathrm{I}^3 - 7.71{*}10^{-5}\mathrm{I}^2 + 1.79{*}10^{-2}\mathrm{I} + 0.49 \quad (22)$$

with “a” being a factor that takes into account the actual number of days in a month (28-31) and the number of daylight hours, the latter being a function of the altitude and the season (Rosenberry *et al*., 2007). The input data (air temperature, T) used in the Thornthwaite approach were collected from 2008 to 2021 and are given in Table 3.8. The air temperature was measured monthly at around 2 meters above the lake’s surface water, while the water temperature was directly measured at the lake’s surface from 2008 to 2021.

Table 3.8: Input data used in the calculations of evaporation from the surface of Lake Baringo, air temperature data coolected from Eldama Ravine online station and literature (Omondi et al., 2014) (2008-2019) and this study (2020-2019).

Month	i	T (°C)	a	d
January	22.75 ± 2.3	25.03 ± 1.2	0.87 ± 0.02	31
February	12.82 ± 1.1	25.39 ± 0.8	0.70 ± 0.03	28-29
March	10.36 ± 1.9	23.67 ± 0.5	0.66 ± 0.13	31
April	11.80 ± 1.7	25.96 ± 1.1	0.68 ± 0.12	30
May	11.08 ± 1.4	24.88 ± 1.5	0.67 ± 0.14	31
June	23.02 ± 2.3	25.33 ± 1.2	0.87 ± 0.12	30
July	22.70 ± 1.8	24.97 ± 2.1	0.86 ± 0.09	31
August	23.35 ± 2.5	25.69 ± 1.6	0.87 ± 0.06	31
September	23.01 ± 2.2	25.31 ± 1.8	0.87 ± 0.07	30
October	21.29 ± 2.4	23.42 ± 1.1	0.84 ± 0.09	31
November	22.40 ± 2.3	24.65 ± 1.3	0.86 ± 0.10	30
December	21.28 ± 2.1	23.41 ± 1.5	0.84 ± 0.02	31
	(I =225.86)			

I: is the annual heat index and represents the sum of monthly i, T is air temperatures in °C, d is the number of days in a month and the parameter “a” is a function of I, *i* is the heat index per month, a is the factor that takes into account the actual number of days (d) in the month (28-31) and the number of daylight hours.

3.3.2.4 Water residence time (R_t)

The water residence time (R_T) was estimated every month in nine sampling stations chosen in the lake from the northern zone to the southern zone for the period from January 2020 to June 2021 by the following equation (Galizia and Matsumura-Tundisi, 2011):

$$R_t = \frac{V}{Q(t)(year)} \tag{23}$$

Where V is the volume of the lake estimated from Obando *et al.* (2016) and Q(t) is the annual total flow (Qt = Precipitation (P) + inflows).

The estimated lake surface (S) obtained from Obando *et al.* (2016) and the monthly depths measured during the data collection period were used to estimate the average volumes of the lake (V) for the sampling period (January 2020-June 2021) using the volume $V = Depth\ (m) * S\ (m^2)$ (Harrilson *et al*., 1994).

3.3.3 Lake level sensitivity analysis

Sensitivity is defined as how the lake responds in relation to variations in its water balance components (precipitation, evaporation, runoff, underground seepage inflow and outflow; etc.) or water levels related to changes in hydrological regimes (Yin and Nicholson, 1998). The sensitivity analysis determines the magnitude of change in each of the required hydrological parameters (air temperature, precipitation, cloud cover, and relative humidity) required to bring the lake level from dry-out to overflow conditions (Mbanguka *et al*., 2016). For sensitivity analysis, monthly meteorological data including air temperature, cloud cover, precipitation, and relative humidity were collected monthly from Eldama Ravine meteorological station from March 2008-June 2021. The annual means were calculated for all the parameters and were used for the water balance modeling and sensitivity analysis. The choice of the annual time frame was motivated by data availability. Furthermore, the annual time frame was appropriate than seasonal dynamics since the study focused on the establishment of an annual water balance of the lake-catchment system and the assessment of the sensitivity to changes in long-term hydroclimatic conditions on an annual basis.

Prior to the sensitivity analysis of the lake level to meteorological variables, the missing water level data for previous years with available lake surface information were derived by using the estimated lake storage capacity (C) curve equation (C = 249.52*Depth – 298.55). The equation was obtained from the lake volume-depth relationship derived using a linear model from January 1970 to June 2021 data. The simple linear model showed a good fit of the lake storage-lake level relationship (Figure 3.3, $R^2 = 0.96$).

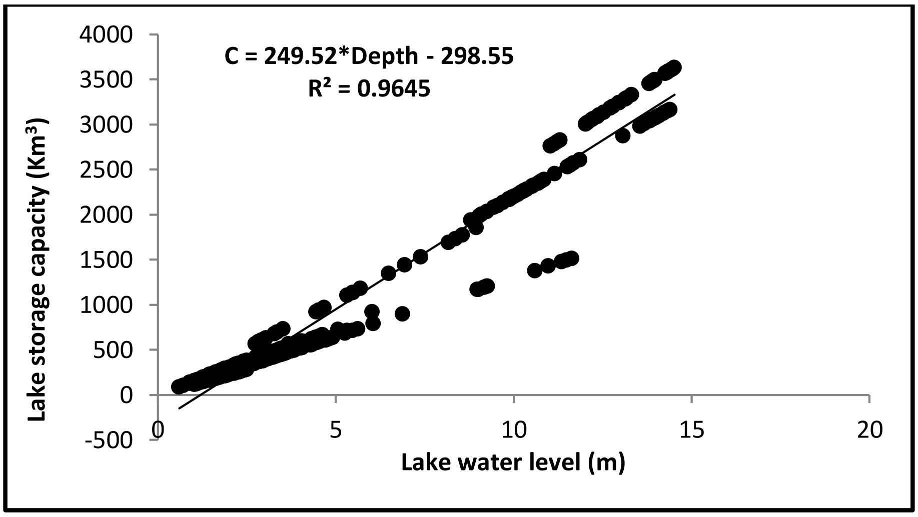

Figure 3.3: Relationship between Lake Baringo storage capacity and water level for the period from January 1970 to June 2021.

The influences of the changes in rainfall, evaporation, air and water temperature, cloud cover, and relative humidity on the annual lake levels were assessed. Multiple scatter regressions were performed between the lake level and each of the selected meteorological variables (rainfall, evaporation, air temperature, cloud cover, and relative humidity) to analyze the sensitivity of the lake level to changes in the parameters as well as to evaluate the effects of the four selected meteorological variables (rainfall, air temperature, cloud cover, and relative humidity on the evaporation from the lake. All the regression analyses were performed using the Sigma Plot statistical package.

3.4 Objective 3: Effects of water level fluctuations on water quality variables and fisheries of the lake

3.4.1 Data collection

Lake water levels (WLs) data were obtained from the Kenya Water Resources Authority (WARA) for the period from 1956-2018. WARA monitors water levels by a graduated wooden scale of 5 m height with the nearest 1-cm accuracy in graduations. The scale was installed at the lowest elevation in the southern part of the lake (Figure 3.1, Table 3.1) to enable monitoring even during low lake levels. Additional water level data were collected through this study from January 2020 to June 2021 using the same sampling methods as WARA. In the year 2019, no data were collected for water levels because the measurement scale got submerged due to rising water levels occasioned by heavy rains. For each year of record, both for the two periods (1956-2018) and (2020-2021), both monthly and yearly mean lake levels were estimated and used to derive the lake water level fluctuation (WLF) indices as described below (see section 3.4.2). Both the water quality variabless and fish landings were measured from three ecological zones (northern, central, and southern) in the lake extending from the northern to the southern parts of the lake (Figure 3.1) in order to cover the representative lake area. The characteristics of the three ecological zones of Lake Baringo are described in Table 3.1.

Long-term rainfall data (1959 to 2021; Table 3.9) collected from 24 meteorological stations installed in the lake's watershed were compiled from the literature (Ngaira, 2006; Obando *et al*., 2016) and the local meteorological services. Meteorological stations are installed in the lake's catchment in both high and lowlands by the Kenyan Water Resource Authority in Rift Valley Basin (Table 3.9).

Table 3.9: Meteorological stations in the catchment of Lake Baringo (Kenya), period of data collection, and mean annual rainfall recorded by the station (Ngaira, 2006; Odada *et al.*, 2006; Okech *et al.*, 2019).

No	Station name	Station ID/Altitude	Data duration	M. A. rainfall (mm yr^{-1})	# of years
1	Elburgon Divisional Forest	9035237	1961-2003	2774	44
2	Mau Summit	9035038	1959-1991	1602	32
3	Molo Water Bailiff	9035266	1967-2001	2115	35
4	Molo Pyrethrum	9035093	1959-1990	2089	32
5	Gatheri Turi	9035099	1959-1975	1083	17
6	Marioshoni Forest	9035117	1959-2003	2772	45
7	Molo Forest	9035273	1959-2003	2250	35
8	Eldama Ravine (ELDR)	Online (from 2009)	1960-2021	959	62
9	Perkerra Station	NA	1968-2000	636	33
10	Tangulbei Station	NA	1960-2000	587	42
11	Station HQ (Omari)	NA	2012-2020	491	9
12	Station F.1	NA	2012-2020	694	9
13	Station F.2	NA	2012-2020	759	9
14	Station F.3	NA	2012-2020	855	9
15	Station F.4	NA	2012-2020	710	9
16	Station F.4a	NA	2012-2020	504	9
17	Station F.5	NA	2012-2020	563	9
18	Station F.6	NA	2012-2020	810	9
19	Station F.7	NA	2012-2020	688	9
20	Station F.9	NA	2012-2020	924	9
21	Station F.10	NA	2012-2020	593	9
22	Station F.11	NA	2012-2020	662	9
23	Station F.13	NA	2012-2020	584	9
24	WARA station	NA	1980-2018	438	38

NA: Not applicable, WARA: Water resource authority, F: Field station, HQ: Headquarter and Nursery station, M.A.: mean annual.

The monthly data on rainfall collected from Eldama Ravine station (ELDR), the station with the longest data series were averaged and compared to the combined average of the rainfall data collected from 23 other local meteorological stations around Lake Baringo for the years from 1959 to 2021. The information was used for the calibration and validation of data collected from Eldama Ravine station. The correlation between the two-time series was done for both monthly mean data and yearly mean data (Figure 3.4a and b) in order to validate the data from Eldama Ravine meteorological station to enable further use of tha data in the lake water balance modeling and lake level sensitivity analysis. The monthly comparisons were based on data collected from 2008 to 2021 while, the annual comparisons were based on data collected from 1959 to 2021 (Table 3.9 and Figure 3.4).

The meteorological data (precipitation, air temperature, relative humidity, and cloudiness) collected from the 24 stations were combined to represent significantly the climatic condition in the lake watershed. Nevertheless, the Eldama Ravine (ELDR) dataset being of reliable satellite products and with the longest dataset in the Rift valley region, this station database was used for meteorological data collection and further analyses in this study due to the good correlation with other stations (Figure 3.4).

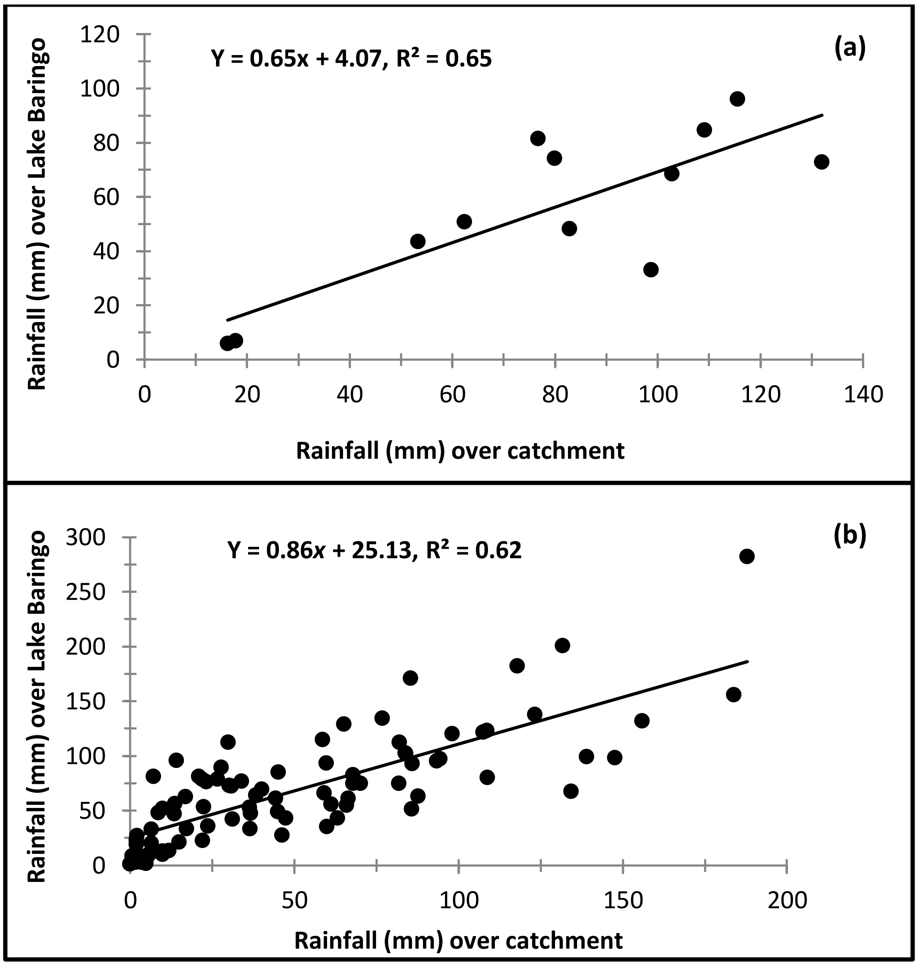

Figure 3.4: Scatter plots of rainfall as derived from satellite station (ELDR) over Lake Baringo vs rainfall measured water over its catchment: (a) relationship of monthly means (2008-2021), and (b) Averages of annual data computed at the 23 stations in the lake catchment for the years 1959-2021 (Table 3.8).

3.4.2 Water Level Fluctuations (WLFs) indices

For both long-term (1956-2020) and short-term (January 2020-June 2021) water level data analyses, two differently time-lagged indices were used as indicators of water level fluctuations in Lake Baringo as proposed by White *et al.* (2008): (1): the difference from the long-term mean (DLTM) (m) was calculated by determining the mean water level from 1956 to 2020 for the lake and then subtracting that mean value (across years) from the mean water level for a particular year. For the short-term analysis, DLTM (m) was calculated by determining the mean water level from January 2020 to June 2021 and then subtracting that mean value (across months) from the mean water level for a particular month. These calculations result in positive and negative values indicating the mean water level for a particular year relative to the lake's overall long-term mean. (2): the annual amplitude (WLamp, m) determined as the difference between the lowest annual (minimum) and highest (maximum) recorded lake level within a year (WLmax-WLmin); provides a measure of the strength of the flood pulse for that particular year. Also, the monthly amplitude (WLam, m) was determined using the same procedure as WLmax-WLmin within a month on daily water level data.

3.4.3 Lake Baringo Fisheries

Long-term data on fish landings (1982-2018) from the lake were obtained from the monthly records of the Kenya Fisheries Department at Baringo Station and the additional short-term data by sampling the landings from January 2020 to June 2021 in this study. The fish landings data used for the two periods (1982-2018) and (January 2020-June 2021) were collected monthly from three major fish landing beaches representative of the lake's fishery, selected based on their accessibility and quantity landed. The three beaches cover the entire lake from the northern to the southern regions and include "Kampi Samaki" in the southern part of the lake, "Ngenyin" in the central zone, and "Komolion" in the northern part of the lake. At each beach, the catch was separated into species before the counting and weighing of the fish. The fish weight was measured by species using a weighing balance (digital scale) while a measuring board was used to measure the fish lengths (standard and total lengths). The data recorded included weight (nearest 0.1 kg), stomach content and length (nearest mm) for each specimen of species landed and randomly sampled from the fishers.

A sample sizes of **92,799** (Table 3.10) were used to calculate the relative condition factors of landed fish used as fisheries performance index. The effects of water level fluctuations on the long-term fish conditions were assessed through regression analysis. The relative condition factor (Kn) was estimated for each species over the period (2008-2020) from the ratio of observed weight to the expected weight-for-length following Le Cren's equation (Le Cren, 1951):

$$K_n = \frac{Wi}{aLi^b} \qquad (24)$$

Where, W_i is an observed individual fish weight and L_i is observed individual fish total length.

The regression constants (a and b) were obtained from the overall length-weight relationship ($W = aL^b$) derived for the species.

The number of fish collected in 2021 was not considered in the long-term analysis but used in the short-term analysis because the fish landings were collected for only 6 months (January to June 2021).

Table 3.10: Number of fish specimens per species used in the estimation of individual relative condition factor for the period 2008-2020 in Lake Baringo, Kenya

#	Fish species/years	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	Total
1	*P. aethiopicus*	1435	2361	509	507	1085	2260	828	327	671	545	1194	5963	5376	23061
2	*B. intermedius*	1539	2444	633	510	1160	2263	814	289	619	848	922	425	5889	18355
3	*C. gariepinus*	1541	2450	631	556	1156	2273	814	328	669	3028	2185	5929	2016	23576
4	*O. niloticus*	1552	2452	652	564	1168	2274	819	333	830	2994	7405	5737	1027	27807
	Total	6067	9707	2425	2137	4569	9070	3275	1277	2789	7415	11706	18054	14308	**92799**

P = *Protopterus*, *B* = *Barbus*, *C* = *Clarias* and *O* = *Oreochromis*

3.5 Objective 4: Modelling the food web properties and fisheries dynamics in Lake Baringo using Ecopath mass-balanced model

The study used the Ecopath component of the Ecopth with Ecosim (EwE) (http://www.ecopath.org) (Christensen, 1995) software to model the ecosystem structure and functioning of Lake Baringo. An overview of EwE model is presented in section 2.3.0. The Ecopath model employs the principle of mass-balance, which assumes that the production of any given preys is equal to the biomass consumed by the predators plus the biomass caught (eg. in fisheries) plus any exports from the system (Equation 25). It also applies the energy flow balance which applies Newton’s second law of thermodynamics (Christensen, 1995). Both the mass-balance and energy balance models are discribed below:

3.5.1 Mass balanced ecosystem model

3.5.1.1 Mass-balance of production

The sum of fishery catches, predation mortality, biomass accumulation, net migration, and other mortality represents the fisheries production (Pi) in the lake and is estimated as follows (Christensen, 1995):

$$Pi = Yi + Bi \times M2i + Ei + BAi + \mathrm{MO}i + Pi \times (1 - EEi) \qquad (25)$$

Where, *Pi* stands for the total production rate of the group (*i*), *Yi* represents the total fishery catch rate of *(i)*, *M2i* is the total predation rate for the group (*i*), *Bi* is the biomass of the group (*i*), *Ei* is the net migration rate (emigration - immigration) and, *Pi* × (1 - *EEi*) is an expression of the "other mortality" rate for the group *(i)*, MO*i*, representing mortality other than caused by predation and fishing. *EE*i, the ecotrophic efficiency, is the proportion of production of the group (*i*) which is incorporated into the next trophic level through predation with the balance being lost through other sources of mortality such as disease, starvation, and old age. *BAi* is the bioaccumulation rate for (*i*) and expresses a positive or negative value indicating changes in population sizes of a fishery or when a group is in decline related to over-fishing (Christensen and Pauly, 1993).

Equation (25) can be rewritten based as below:

$$Bi\ x\ \left(\frac{P}{B}\right)i - \sum_{j=1}^{n} Bj\ x\ \left(\frac{Q}{B}\right)j\ x\ DCji - \left(\frac{P}{B}\right)i\ x\ Bi\ x\ (1 - EEi) - Yi - Ei - BAi = 0 \qquad (26)$$

Where for each functional group: *P/B* is the production/biomass ratio also equal to total mortality Z. yr^{-1}, *Q/B* is the consumption per unit of predation j biomass, and *DCji* is the fraction of prey (*i*) in the average diet of predator (*j*).

3.5.1.2 Energy flow balance

In the mass-balanced model (Equation 26), the Energy input and output of all biomass compartments must be balanced, such that (Christensen *et al*., 2000; Darwalla *et al*., 2010):

Consumption = production + respiration + unassimilated food (27)

Respiration (a parameter rarely measured in fisheries analysis) is estimated as the difference between consumption and production (plus unassimilated food) which are parameters commonly estimated in fisheries analyses. Ecopath with Ecosim (EwE) software uses six basic parameters (Natungonza *et al*., 2016) of which at least four input parameters (B, P/B, Q/B, and EE) are required to estimate the missing parameters and to estimate energy flows and standing biomasses between functional groups based on equations (26), and (27) (Pauly *et al*., 2000). EE is oftenly estimated automatically by the Ecopath model (Christensen, 1995).

3.5.2 Data collection and model set-up

Three annual Ecopath mass-balance models (1999, 2010, 2020) were constructed for the Lake Baringo ecosystem and temporal trends in the parameters compared. The 1999 and 2010 models were based on secondary data collected from the literature (Aloo, 2002; Ballot *et al*., 2003; Mlewa, 2003; Terras-Wahlberg *et al*., 2003; Mlewa *et al*., 2006; Odada *et al*., 2006; Omondo *et al*., 2014; Aura *et al*., 2020) and unpublished reports from Kenya Marine and Fisheries Research Institute (KMFRI) and Kenya Fisheries Department. The third Ecopath model (2020) was built based on this study's sampling for the year 2020. All models were run using the Ecopath with Ecosim (EwE) Software 6.6 (Christensen and Walters, 2004). The models were fit for the trophically linked biomass pools and concentrated on the major system biomass components (Walters *et al*., 1997; Pauly *et al*., 2000; Christensen *et al*., 2005). Biomass groups used in the model were defined as species or functional groups (biota performing the same function or of similar traits) quantified as wet weights. Each of the three models represented the annual average condition of Lake Baringo's ecosystem and its functioning. For comparison reasons at the system level, the three ecosystem models were constructed with the same number of functional groups (n = 8) to reduce bias associated with different aggregation strategies (Gaichas *et al*., 2009). The current study did not include aquatic birds and mammal (hippos) and reptiles (crocodiles) in the models due to a lack of scientific information (biomass, diet, biological measurements) on the groups in Lake Baringo.

3.5.3 Functional groups used in the model

Eight functional groups were identified in the present study for use in the model, including four (4) fish species (*0reochromis niloticus baringoensis, Protopterus aethiopicus*, *Clarias gariepinus*, and *Labeobarbus intermedius*). *Labeo cylindric* was not considered in this study because of lack of information, the fish being rare in the landings, and being of less economic importance. The fish species were selected based on the availability of reports on the fish catch data for the lake and their economic importance (KMFRI Technical Reports 2017, 2018; Fisheries Department Technical Reports, 2015, 2016, 2018). Further, four other groups constituting important functional groups included; mollusks, zooplankton, phytoplankton, and detritus made the remaining functional groups.

3.5.4 Fish landing catches used in the model

Data on fish landings for the 1999 and 2010 models were collected from the monthly records of the Kenya Fisheries Department at Baringo Station and different literature sources (Aloo 2002; Mlewa, 2003; Mlewa *et al*., 2006; Odada *et al*., 2006; Omondi *et al*., 2014; Aura *et al*., 2020). The landings data used in the 2020 model were collected from January 2020 to December 2020 in this study. The fish landings data used in the three annual models (1999, 2010, 2020) were collected monthly from the same three major fish landing beaches selected based on their accessibility and quantity landed (see section 3.4.4; Table 3.9). The annual fish yields (kg) were used for the biomass calculations in the model using EwE software 6.6 (Christensen *et al*., 2005). The data on annual landings, total length, number of specimens for each fish species (n), and related estimated variables (L_{∞}, L_{mean}, L_c and L_{max}) used as inputs in the three models of Lake Baringo are presented in section 4.4.3 (Table 4.15).

3.5.5 Estimation of length-based parameters used in the models

The individual total length (TL) and the determination of the length at first capture (*L*c), mean length (*L*mean), and maximum length (*L*max) for each species of the four fish species (*Protopterus aethiopicus, Oreochromis niloticus, Clarias gariepinus,* and *Labeobarbus intermedius*) used in the models (See section 4.4.3; Table 4.15) were obtained after each monthly sampling for the years 1999, 2010 (Aloo 2002; Mlewa, 2003; Mlewa *et al*., 2006; Odada *et al*., 2006; Omondi *et al*., 2014; Aura *et al*., 2020) and 2020 (this study). For the three Ecopath models, a total of 18,237 fish specimens were used for the determination of the different length categories (Lmax, Lmean, Lc) and included 6,338 specimens of *P. aethiopicus;* 6,781 specimens of *L. intermedius;* 3,105 specimens of *C. gariepinus,* and 2,014 specimens of *O. niloticus baringoensis*

*(*See section 4.4.3; Table S3). The empirical formula described by Froese and Binohlan (2000) was used to estimate the asymptotic length ($L\infty$), as follows:

$$\log L\infty = 0.044 + 0.9841 \times \log (L\text{max}) \qquad (28)$$

The total mortality (Z), also equal to P/B, was then calculated using Beverton and Holt's equation (Beverton and Holt, 1957) incorporated in FiSAT Software (Gayanilo *et al.*, 1996) as follows:

$$Z = k \times \frac{(L\infty - L\text{mean})}{(L\text{mean} - L\text{c})} \qquad (29)$$

Where, k = growth coefficient calculated using FiSAT packages.

3.5.6 Diet composition per group

The diet composition required in the Ecopath model is a key information in the dynamics of ecosystems. The connection of food networks between different ecological groups (Fetahi and Mengistou, 2007) in Lake Baringo were estimated by the stomach content analysis. The monthly data samples collected in the year 2020 were used for diet composition analysis for the 4 commercial species. While, for 1999 and 2010 secondary data on the diet composition of the species were obtained from different literature sources (Oduor *et al*., 2003; Schagerl and Oduor, 2003; Tarras-Wahlberg *et al*., 2003; Odada *et al*., 2006) and KMFRI unpublished monthly reports (KMFRI technical reports 2017, 2018). The stomach content composition of the species (DCJi, equation 30) was then used in the three Ecopath models of Lake Baringo. The diet composition information for the four commercial species and other functional groups used in the Ecopath model of the lake is summarized in Table 3.11.

Table 3.11: Diet composition matrix (in % of the weight of the stomach content) of consumers used in Ecopath model for Lake Baringo, Kenya

Groups/species name	1	2	3	4	5	6	7	8
1-*P. aethiopicus*		-	-	-	94.3[a,d]	-	-	5.7 [a,c,d]
2-*B. intermedius*		-	-	-	8.8[b]	-	19.7 [b]	71.5[b,d]
3-*C. gariepinus*		-	-	-	49.2[a]	19.6[a]	2.9[a]	28.3[a]
4-*O. niloticus*		-	-	-	-	12[a]	88[a]	-
5-Mollusks				-	-	50[d,e]	50[d,e]	-
6-Zooplankton		-	-	-	-	-	100[a,b,d]	-
7-Phytoplankton		-	-	-	-	-	-	-
8-Detritus		-	-	-	-	-	-	-

(a) Omondi *et al.* (2014); (b) Odhiambo and Osure (2021); (c) Mlewa (2003); (d) this study. Species names are as in section 3.5.3.

3.5.7 Basic Input parameters used in the Ecopath models

Four main basic inputs required in EwE (Christensen, 1995) were used in each constructed Ecopth model including; biomass (B) in t km^{-2}, production/biomass ratio (P/B) per year, consumption/biomass ratio (Q/B) per year and ecotrophic efficiency (EE). For each functional group, three of the four parameters (B, P/B, Q/B, and EE) were estimated and entered as model inputs. The missing parameter (EE) was estimated by the Ecopath parameterization routine (Christensen *et al*., 2005; Christensen and Walters, 2004). EE is the most difficult one to estimate and is thus often left unknown in Ecopath models (Christensen, 1995).

3.5.7.1 Fish species biomass (B)

The biomass (*B*) is the total mass per functional group expressed as metric ton/km^2. The biomass per habitat area of each fish species and time period (1999, 2010, 2020) was calculated using the equation $B = Y/F$ (Gulland, 1971) assuming F = 0.5 x Z (Rehren *et al*., 2018), where *Y* is the fish catch and *F* is the fishing mortality coefficient obtained from FishBase for each species and rationalized with estimates in other Kenyan lakes (Njiru *et al*., 2018). All fish biomasses were expressed in wet weight.

3.5.7.2 Production/biomass ratio (P/B)

Production/biomass ratios are difficult to estimate directly but the ratio is equal to total mortality (Z) (Pauly *et al*., 2000). The P/B ratio of fish groups presents the steady state of the ecosystem (Allen, 1971) and is equivalent to the instantaneous rate of natural total mortality (Z) (Pauly *et al*., 2000). Total mortality (Z) of exploited fish species was determined by summing the value of fishing mortality (F) and natural mortality (M) as Z = F + M.

In this study, the Production/ Biomass ratio (*P/B*) was therefore assumed to be equal to the instantaneous mortality (*Z)* estimated for all fish species using the length-converted catch curve routine incorporated in FiSAT software (Gayalino *et al*., 1996).

3.5.7.3 Consumption/biomass ratio (Q/B)

Q/B was estimated for each consumer functional group using based on the relationship proposed by Palomares and Pauly (1998) as follows:

$$\text{Log}(Q/B) = 7.964 - 0.204 \log W\infty - 1.965T + 0.083A + 0.532h + 0.398d \quad (30)$$

Where, W∞ (W∞ $= qL\infty^3$is the asymptotic weight (g) equivalent of L∞; T is an expression for the mean annual temperature of the water body, expressed using T = $\frac{1000}{K}$, (K = °C + 273.15);

A is the aspect ratio_(A= $\frac{hc^2}{s}$) of the caudal fin of the fish, giving the height of the caudal fin (A approximately equals to 1.32 and 1.9 for fish with round and forked tails) (Froese and Pauly, 2016), (hc) and surface area of the caudal fin (s); h is a dummy variable defining the type of food (1 for herbivores and 0 for detritivores and carnivores), and d is also a dummy variable expressing food type (1 for detritivores and 0 for herbivores and carnivores). A mean annual water temperature was determined for each year from literature and this study and used in the model.

3.5.8 Ecological functioning and fisheries indicator outputs

After the models were balanced, the ecosystem stability and system maturity (indexed by P/R) were assessed by different network statistics and flow indices based on Odum's (1969) and Christensen's (1995) approaches. Then, the modeled structure and function of lake ecosystem were compared during the three time periods (1999, 2010, 2020) using various calculated network and information flow indices (Christensen, 1995) as below:

(1) Ecotrophic efficiency (EE) is the parameter used to define a balanced mass model.

EE is defined as a fraction of the production of an ecological functional group (species) consumed by another ecological group in the system or caught by fishing activities (Dutta *et al.*, 2017) and varies between 0 (relating to less exploitation of a group in the systems) and 1 (relating to exploited systems) (Christensen *et al.*, 2000).

EE value was an output estimated (auto-estimated) by Ecopath with Ecosim software based on *B, P/B,* and *Q/B,* the key input parameters. EE was used in this study to estimate the exploitation rate of fish species and the need for formulating conservation measures for a sustainable fishery in the lake if necessary.

(2) Network flow indices used in the present study: included the total system respiration

(TR), total production (P), total consumption (Q), total export (EXP), total primary production (TPP), net system production (NSP), total primary production/total production ratio (TPP/TP), total primary production/total respiration ration (TPP/TR) and total flow to detritus (FD) represented by the total system throughput (TST).

TST is a quantitative and summative variable that determines both the size of the entire system based on flow and all biomass flows within an ecosystem (Ulanowicz, 1986). The TST is defined as the sum of all flows in a system. It represents the "size of the entire system in terms of flow" (Ulanowicz, 1986). As such, it is an important parameter for comparisons of flow-networks and describing network lengths between models.

Other indexes included the sum of all system production (P), and net primary production (NPP), which provide an index of activity in the ecosystem associated with the first trophic level. The total biomass without detritus (B), mean trophic level of catch (TL), and total catch (Ca). Odum (1969) and Finn (1976) indexes (total primary production/total respiration (TPP/R), total primary production/total biomass (TPP/B), biomass supported per unit of energy flow (B/TST), net system production (P), connectivity index (CI), omnivory index (OI), average path length (APL) and Finn's cycling index, FCI) were also used to evaluate the ecosystem maturity and development stage. The transfer efficiency (TE) of each trophic level (TL) was calculated as the ratio of the summed exports from a given TL plus the flow transferred from one TL to the next TL indicating the efficiency of a transfer from TL_{n-1} to TL_n (Lindeman, 1942). The ratio of total system biomass to the total system throughput (*B*/TST) (Christensen, 1995) is directly proportional to system maturity where the estimated value tends to be low during the ecosystem development phase and increases as a function indicating the maturity stage of the ecosystem (Odum, 1971; Ulanowicz, 1986). This index was used to assess the maturity stage of the lake in this study.

The Net Primary Production to Total Respiration (NPP/TR) ratio is another system maturity index (Odum, 1969; Pérez-España and Arreguín-Sánchez, 1999) with values close to 1 indicating mature ecosystems. The net production of the system defined as the difference between primary production and total respiration (TPP-TR) is another index of system maturity (Odum, 1969) and equals to zero in a truly balanced ecosystem.

The System Omnivory Index (*SOI*) is the average omnivory index of all consumers weighted by the logarithm of each consumer's food intake in the system (Christensen *et al*., 2000). It indicates the diversity or complexity of food webs (Christensen, 1995).

The connectance index (*CI*) (Odum, 1969) for a given food web is computed as the ratio of the number of actual links between groups to the number of theoretically possible links. Feeding on detritus (by detritivores) is included in the count, but the opposite links (i.e., detritus 'feeding' on other groups) are disregarded. This index was also used to assess the maturity of the lake because it presents a significant correlation with the maturity of the ecosystem suggesting that a food chain structure changes from linear to web-like (high CI) as a system matures (Odum, 1969; 1971).

Finn's cycling index (FCI) (Finn, 1976) is the proportion of the total system throughput (TST) recycled in the system. According to Monaco and Ulanowicz (1997), cycling is considered to be an important indicator of an ecosystem's ability to maintain its structure and integrity through positive feedback and can be used as an indicator of stress (Ulanowicz, 1986) or system maturity (Christensen, 1995; Vasconcellos *et al*., 1997). This index was used to evaluate both maturity and stress rate (diminishing FCI) of the Lake Baringo ecosystem.

(3) Informative flow indices are commonly used to evaluate the food web robustness or the fragility of a food chain in an ecosystem (Canning and Death, 2021).

They include: flow uncertainty metric (H), Ascendency flowbits (A), relative Ascendency (A/C), overhead (Ov), overhead and capacity ratio (Ov/C), redundancy system entropy (RSE), capacity flowbits (C) and average mutual information (AMI). These indices were used in this study to evaluate the fragility of Lake Baringo's food web. In this study, the used flow indices included only: Ascendency (A) to measure both the degree to which energy flows are confined to specific pathways and the energetic size of the lake's web (Ulanowicz, 1997b). The Ascendency being also a measure of system growth (i.e. age, size) and Development (i.e. organization) of network link, it was used to provide the same information for the lake. Another informative flow index used in this study was the system overhead (Ov) flowbits. The fraction of a system's capacity not considered as Ascendency is considered as the Systems Overhead (Ov), which is considered as the energy in reserve of an ecosystem

(Monaco and Ulanowicz, 1997), the resilience capacity of a system, especially in the case of perturbations(Ulanowicz, 1986). It was used in this study to evalution the rate of perturbation of the lake over time. Flow indices such as Ascendency and Overhead are used to assess the stability and maturity of an ecosystem (Christensen, 1995). Ascendency is, therefore, a measure of average mutual information in a system (Ulanowicz and Norden, 1990).

(4) Assessing the role of fisheries of Lake Baringo from model outputs

Three fisheries indices (indicators of fishing intensity and impacts in the ecosystems) were computed to identify the current state of fisheries of Lake Baringo. The fisheries indicators included the mean trophic level of the catch (TLc), the primary production over total ecosystem production known as "primary production required" (PPR) and the gross fishing efficiency (GFE) were also used in this study to evaluate the fisheries dynamics of Lake Baringo over time.

i) Mean trophic level of the catch (TLc)

This indicator reflects the overall strategy of a fishery, and is calculated by weighting the proportions of each fish/fish group from the catch by their respective trophic levels (TLs). The TLc decreases as fishing impacts increase in the ecosystem since fishing tends to first remove the higher TL organisms.

The mean trophic level of the catches (TLc) was calculated as the weighted average of trophic level (TL*i*) of caught species using catches (Y*i*) as the weighting factors (Pauly *et al*., 1998):

$$\mathbf{TLc} = \frac{\sum_i \mathbf{TL}\boldsymbol{i}.\mathbf{Y}\boldsymbol{i}}{\sum_i \mathbf{Y}\boldsymbol{i}} \qquad \textbf{(31)}$$

ii) Primary production required (PPR)

The PPR is the primary production that is necessary to sustain the catches, and estimated from both primary producers and detritus in order to assess the sustainability of fisheries in terms of energy. It is empirically calculated using a formula suggested by Pauly and Christensen (Pauly and Christensen, 1995) that links PPR, fish catches (Yi), mean transfer efficiency (TE), and trophic level (TL):

$$PPR = \frac{1}{9} . \sum_i Yi . \left(\frac{1}{TE}\right)^{TLi-1} \quad (32)$$

The mean transfer efficiency (TE) for the food web was calculated as the geometric mean of transfer efficiencies for each of the integer trophic levels II to III. The transfer efficiency of a trophic level is calculated as the sum of the flow transferred from any given level to the next higher level, plus exports (e.g., catches) from the given level relative to the input (or throughput) of the given trophic level (Pauly and Christensen, 1995).

iii) Gross fishing efficiency (GFE)

This indicator is computed as the sum of all fisheries catches divided by primary production (PP) (Actual catch/Primary production). The actual ratio indicates high values for overerexploited systems or characterized by more efficient use of the system's production (i.e. harvesting fish low in the food web), and low values in systems whose fish stocks are underexploited, or where the fishery is concentrated on apex predators (Pauly *et al*., 1998; Garcia *et al*., 2012).

3.5.9 Keystoneness Index (KSi)

The keystoneness index (KSi) was calculated for the three trophic models in Lake Baringo to quantitatively assess the impact of each of the functional groups on other groups' biomass in the system affecting the ecosystem functioning (Libralato *et al*., 2006). A keystone species is considered as the group that would significantly affect other functional groups even with relatively small biomass. Functional groups with low biomass and high overall effect are attributed to high keystone values (Christensen *et al*., 2008; Abobi *et al*., 2021). The keystoneness was estimated in this study based on the incorporated Ecopath equation based on Libralato *et al.* formula (Libralato *et al.*, 2006).

$$KS_i = log[\varepsilon i(1 - pi)] \quad (33)$$

Where, KS_i is the keystoneness of group i, ε_i is the overall effect of group i and pi is the proportional biomass of group i.

3.5.10 Prebalancing conditions (PREBAL) of Ecopath model

Prebalancing (PREBAL) diagnostics evaluates if the data used in the trophic model are realistic or coherent to the system level based on some basic laws, rules, and principles of ecosystem ecology (Link, 2010). Prior to the balancing of the three trophic models (see section 3.5.8), PREBAL diagnostics were used to evaluate the Ecopath model assumptions in the initial model before progressing to dynamic simulations as suggested by Christensen (1995).

In this study, after the initial Ecopath model was created, a number of pre-balance factors were used. They include assessing patterns of biomass across taxa/trophic levels (where biomass should decline on a log scale across trophic levels indicating fewer or lesser individuals at higher trophic levels); biomass ratios (where predators biomass should be less than of 1 relative to their prey); and vital rates across taxa/trophic levels (these should be a general decline with increasing trophic level) (Christensen, 1995).

3.5.11 Balancing and calibration of the models

After Prebal diagnostics confirmed in used assumptions, the Ecopath model is then balanced.

A model is mass-balanced if the catches, consumption, biomass accumulation, and export do not exceed the production of a group (Equation 26) (Polovina, 1984; Ullah *et al*., 2012). The model is considered balanced when it is characterized by an EE value less than 1 (Ullah *et al*., 2012) and the P/Q ratio between 0.1 and 0.3, and the expected range for *P*/*Q* varies according to the organism (Christensen *et al*., 2000; Christensen *et al*., 2004). EE value exceeding 1 suggests a very high demand for a particular functional group in the ecosystem (Mohamed *et al*., 2008). During balancing of the models in this study, the diet composition and mean biomass were modified until the achievement of a balanced model (e.g. EE < 1) while, the mean biomass was slightly adjusted to within 1 standard deviation (SD) of its original (average) value (Mohamed *et al*., 2008). The final P/B values of the different functional groups were modified up to between 10-15% of their initial magnitudes for the balancing of the model according to Christensen *et al*., (2008).

3.5.12 Pedigree Index (P)

The Pedigree is a measure of the goodness of fit of the model and is calculated according to an equation incorporated in the Ecopath model (Christensen *et al*., 2000). The pedigree index (P), used in this study as a measure of the reliability of the input data origin and of likely uncertainties associated with the input data (reliability of data) (Christensen *et al*., 2000). The Pedigree Index was derived as (Christensen *et al*., 2000):

$$P = \sum_{i=1}^{n} \left(\frac{I\ ij}{n}\right) \qquad (34)$$

Where, Iij is the pedigree index value for group i and parameter j for each of the n living groups in the ecosystem; j can either represent B, P/B, Q/B, Y (yields or catches), or the diet.

The pedigree (P) scales range between 0 and 1 (inclusive). The P index values scale from 0 for data input that are not rooted in local data up to 1 for data that are fully-rooted in local data indicating a good fit of the model (Christensen *et al*., 2000).

3.6 General Data treatment and Statistical Analyses

The water quality variable values (section 4.1.1) were assessed using the water quality index (WQI) as an indicator of the suitability of the lake water for both human consumption and multipurpose uses (WHO, 2008), while organic pollution index (OPI) (section 4.1.3) was used as an estimate of the organic pollution level in the lake (Leclercq and Maquet, 1987). The WQI was estimated using the weighted arithmetic method (Brown *et al*., 1972). The calculated WQI values were compared to various recommended international standard limits (WHO, 2008, 2011). Two-way analysis of variance (ANOVA Two-Factor) was used to determine the influence of both sampling stations and years (long-term scale) as well as the sampling stations and months (short-time scale) on the water quality variables in the lake. The mean values of the physico-chemical variables that showed significant differences between the sampling stations (p < 0.05) were then compared using one-way ANOVA on log-transformed data combined for the study years. Because the year effects were significant (see Table S1; appendex I), the temporal variation of the selected physico-chemical variables (temperature, conductivity, hardness, Secchi disc, TDS, TSS, Chl-a, turbidity, alkalinity, fluoride, SiO_4^{4-}, pH, total phosphorus, PO_4^{3-}, NO_3^-, NO_2^-, NH_4^+, total nitrogen and DO) for the years 2008-2020 were plotted and smoothed trend lines were fitted to the data series using a Locally Weighted-Scatterplot Smoother (LOWESS; Cleveland, 1979) in the MINITAB statistical package. The LOWESS's assumption is based on a weighted least-squares algorithm that partitions local weights with the most influence, while also minimizing the effects of outliers. A smoothness parameter (*f*) of 0.2 was found to adequately smooth the data without distorting the temporal patterns. The co-variation among the physico-chemical parameters, including the WQI, was tested using Pearson's linear correlation. Pearson's linear correlation and simple linear analyses were carried out using the PAST 32.6b statistical package (section 4.1.2).

The effects of meteorological variables on the lake water levels were evaluated by plotting the annual values of each variable against the mean annual lake level. On each plot, the present value of the lake level with its corresponding value of the meteorological parameter was plotted, and the lake level's sensitivity was described as its changes (responses) compared to the current value (section 4.2.1).

Simple multiple linear regressions were performed between the lake level and each of the selected meteorological variables: air temperature, precipitation rate, relative humidity, and cloud cover to analyze the sensitivity of the lake to climate variability. All the regression analyses were performed in the Sigma Plot package. After modeling the dependence of lake level to the variables and testing the sensitivity of the lake level to the meteorological variables, the relationships between the evaporation rate (on of the major water balance components of the lake) in the lake and some climate parameters: air temperature (ta), precipitation (P), cloud cover fraction (cc) and relative humidity (rh) were tested by changing arbitrarily but independently their values by ± 10% (Vallet-Coulomb, *et al*., 2001) to explore their influence on the final lake water balance modeling results (Dühnforth *et al*., 2006).

Principal components analysis (PCA) was used to group the years (2008-2020) based on the similarity of physico-chemical variables and water level fluctuations (see section 4.3.1). Linear regression (y = ax + c) and waveform sine regression (y = a*sin (2*p*x/b + c) analyses (Zar, 2010) were performed to determine the best-fitting model to explain the patterns of lake level fluctuations (WLFs) over the years. Where both models were not significant, a Locally-Weighted Scatter Plot Smoother (LOWESS, Cleveland, 1979) was used to describe the pattern of lake level fluctuations (see section 4.3.2).

Pearson's correlation was performed to determine the concordance between WLF indicators (DLTM and Amplitude, WLamp) and fisheries landings and with water quality parameters (conductivity, turbidity, chlorophyll-a, DO, temperature, TP, PO_4^{3-}, NO_3^-, TN, SiO_4^{4-}, NH_4^+ and WQI). Both Pearson's correlation and linear regression analyses were conducted on log (x + 1) transformed data to meet the required assumption of normality of the dataset (Zar, 2010). The depth variation of the lake during the periods from 1956 to 2021 was determined from a histogram of pixel depths while the lake water level – lake surface area relationship was determined using a linear regression model (see section 4.3.2).

Linear regression and non-linear Gaussian distribution (Zar, 2010) were used to model the influence of WLFs on the lake's fishery yields and fish conditions (a measure of growth performance). The Gaussian distribution (see section 4.3.3) followed a unimodal pattern and tested the hypothesis that the lake fisheries production and fish condition will correspond to optimum WLFs levels below and above which a decline is realized (Lukas, 1942, Keto et al., 2006). The fish relative condition factor (Kn) was estimated for each species over the period 2008-2020 from the ratio of observed weight to expected weight-for-length following Le Cren's equation (Le Cren, 1951). All the graphical plots and major analyses were implemented in the Sigma Plot software.

CHAPTER FOUR

RESULTS

4.1 Objective 1: Spatio-temporal variations in selected water quality parameters and lake trophic status

4.1.1 Physico-chemical water quality variables and Quality standards

The long-term (inter-annual) data analysis (2008-2020) showed significant influences of station and year of sampling on all the physico-chemical and nutrient variables in the lake except for turbidity that did not show significant differences among stations at inter-annual temporal scale (Table S1 in appendex II). In contrast, the influences of station and month of sampling on the physico-chemical parameters at intra-annual temporal scale (2020-2021) were not significant except for DO, Secchi disc, TN, and TDS that showed significant differences among stations (Table S2 in appendex II).

The interaction between years and stations (Two-way ANOVA results) in influencing pysico-chemical prameters was significant for all the measured variables except for four (turbidity, Secchi disc, chlorophyll-a, and TN) (Table S1, appendex II). Conversely, there was no significant interaction between months and stations for all the measured variables for the short-term scale (2020-2021) in the lake (Table S2, appendex II). The lake water quality variables and their international standard limits for both human consumption and aquatic life conditions are shown in Table 4.1 for long-term data (2008-2020).

Table 4.1: Statistical summary of annual means of some physico-chemical parameters of Lake Baringo for the period 2008-2020, Kenya. Bold figures are significantly different between stations or were above WHO/APHA thresholds. ANOVA results are significant at p < 0.05.

	Sampling stations								
	S2	C1	C2	C3	N2	ANOVA		Values criteria	
Parameters	Mean ± SD	Mean ± SD	Mean ± SD	Mean ± SD	Mean ± SD	F	P	WHO (2008)	**
DO (mg/L)	**5.73±0.92**	**6.42±0.94**	**6.43±0.95**	**6.14±0.84**	**6.48±0.86**	8.84	**0.001**	5	6 [1,2,3,4]
Temp. (°C)	**24.94±1.42**	**26.67±2.99**	**26.43±1.63**	**25.61±1.53**	**26.33±1.47**	**21.23**	**0.001**	< 30	-
EC (µS/cm)	509.87±128.99	513.57±132.98	526.01±138.50	518.34±143.18	524.40±137.51	0.13	0.97	1000	1500 [3]
pH	8.356±0.48	8.41±0.50	8.45±0.52	8.42±0.50	8.43±0.48	0.36	0.836	6.5-8.5	6.5-9 [1,2,3]
TUR (NTU)	**56.08±56.36**	**51.46±43.79**	**64.43±111.40**	**59.40±56.46**	**49.75±42.96**	0.47	0.755	< 5	5 [1]
Secchi (cm)	60.74±41.75	65.84±42.66	66.72±43.71	62.90±42.75	69.52±45.49	1.12	0.348	-	-
HD (mg/L)	60.64±18.87	59.31±18.75	59.68±20.50	61.01±19.35	59.95±17.86	0.09	0.985	100-300	500 [1]
Alk.(mg/L)	175.21±41.25	179.97±46.04	179.29±44.59	176.19±44.59	178.89±45.83	0.19	0.944	500	≥ 20 [4]
TDS (mg/L)	235.49±91.36	224.10±103.26	229.88±102.64	226.05±104.48	230.07±102.95	0.11	0.981	< 500	500 [1]
F^- (mg/L)	**6.62±5.20**	**6.91±5.12**	**6.94±5.04**	**7.59±5.75**	**6.82±5.18**	0.17	0.953	1.5	5 [1]
PO_4^{3-} (µg/L)	24.10±13.11	22.20±16.76	23.03±16.76	20.85±9.98	18.42±12.78	0.31	0.869	< 30	< 100 [1,4]
TP (µg/L)	**122.75±87.51**	**121.41±75.13**	**129.34±84.68**	**125.47±84.98**	**134.28±87.15**	0.08	0.987	70	50 [4,5,6]
NO_2^- (µg/L)	**5.14±2.09**	**9.71±9.57**	**7.01±3.77**	**3.90±2.12**	**4.51±3.33**	**2.86**	**0.031**	300	< 100 [1,4]
NO_3^- (µg/L)	9.62±2.91	13.10±9.45	10.59±5.09	9.28±3.40	8.77±4.09	1.28	0.286	1000	< 1000 [1,4]
TN (µg/L)	291.28±179.14	375.34±243.33	360.90±166.56	422.78±213.66	379.68±207.38	0.37	0.827	5000	4000 [5,6]
SiO_4^{4-} (mg/L)	**18.70±2.53**	**21.21±5.30**	**21.57±4.49**	**18.35±3.31**	**21.56±4.44**	**2.34**	0.064	5	-
NH_4^+ (µg/L)	31.43±15.45	20.94±13.14	32.47±16.64	25.46±4.41	30.34±7.96	2.25	0.074	< 1500	< 1000 [1,4]
Chl a (µg/L)	4.33±2.54	8.10±13.52	5.72±4.17	5.94±7.60	4.13±2.03	1.15	0.341	-	12 [2,3]
TSS (mg/L)	14.94±5.05	12.94±5.58	12.07±4.63	10.92±4.39	12.18±4.45	1.23	0.307	-	< 30 [1,2,3,4]
Zeu (m)	1.97±1.27	2.07±1.32	2.08±1.34	1.87±1.30	2.17±1.40			-	-

****** Standard limits for aquatic fauna: **1**. CCME (1999), **2**. APHA (2005), **3**. Rodier *et al.* (2009), **4**. ANZECC (2000), **5**. WHO (2011) and **6.** CSST (1997) (2008). HD: Hardness; Alk.: Alkalinity; TUR: Turdibity; EC. Electrical conductivity; DO: Dissolved oxygen; TDS: Total dissolved solid; TSS: Total suspended solid; TP: total phosphorus; TN: total nitrogen, Chl.a: chlorophyll-a, Zeu: euphotic zone.

However, the means (± SD) of three parameters (DO, Temperature and NO_2^-) showed significant variations ($p < 0.05$) among five stations in the lake (Table 4.1). Dissolved oxygen (DO) was very high in the northern station N2 (6.48 ± 0.86 mg L^{-1}) and its lower value was obtained in the southern station S2 (5.73 ± 0.92 mg L^{-1}) while, temperature showed higher mean values in the central station C1 (26.67 ± 2.99 °C) and lowest values were obtained in the southern station S2 (24.94 ± 1.42 °C). NO_2^- was significantly different ($p < 0.05$) between the stations with higher values in the central lake station C1 (9.71 ± 9.57 µg L^{-1}) and lowest in the near-shoreline station C3 (3.90 ± 2.12 µg L^{-1}). Although all the other parameters did not show significant variation between stations ($p > 0.05$), some of them had values that were above the WHO recommended thresholds for human consumption and ecological integrity. For example, Fluoride (F^-) levels in the lake waters varied from 6.62 ± 5.20 mg L^{-1} to 7.59 ± 5.75 mg L^{-1}, and is well above the WHO permissible level of 1.5 mg L^{-1}. Similarly, turbidity levels of the lake water varied from 49.75 ± 42.96 NTU at northern station N2 to 64.43 ± 111.40 NTU at central station C2, and these were also above the WHO recommended maximum values of 5 NTU for human use (Table 4.1).

Total nitrogen (TN) contents in the lake shifted from 291.28 ± 179.14 µg L^{-1} at southern station S2 to 422.78 ± 213.66 µg L^{-1} at the central station C3. The productivity of the lake, as measured by Chlorophyll-a values, varied from 4.13 ± 2.03 µg L^{-1} at northern station N2 to 8.10 ± 13.52 µg L^{-1} at central station C1 and did not vary significantly ($p = 0.341$) between the stations indicating uniform productivity within the lake.

Three physico-chemical variables (fluoride, turbidity, and chlorophyll-a, Chl-a) fluctuated significantly at annual temporal scale, while others fluctuated slightly over the years (Figure 4.1). Hence, Secchi depth, a measure of lake's water transparency exhibited increasing trends from 2012 to 2016 with a subsequent decline to 2020. Chl-a, a measure of lake's productivity, was low and uniform during 2008-2018 with a subsequent rise in productivity during 2018-2020 (Figure 4.1). Fluoride levels have important health implications; the values peaked in 2009 and 2013 and have subsequently declined to low values from 2016 to 2020. The levels of dissolved oxygen in the lake have fluctuated over the years with a peak value in 2010 (~ 8.6 mg L^{-1}) that subsequently declined to a low of ~5.4 mg L^{-1} in 2020. Conductivity (a measure of water quality degradation) peaked at 769. 5 µs cm^{-1} in 2012 and then declined to a low of 409.4 µs cm^{-1} in 2018 while, total suspended solids (TSS) showed a pattern of increasing values from 2016 (Figure 4.1).

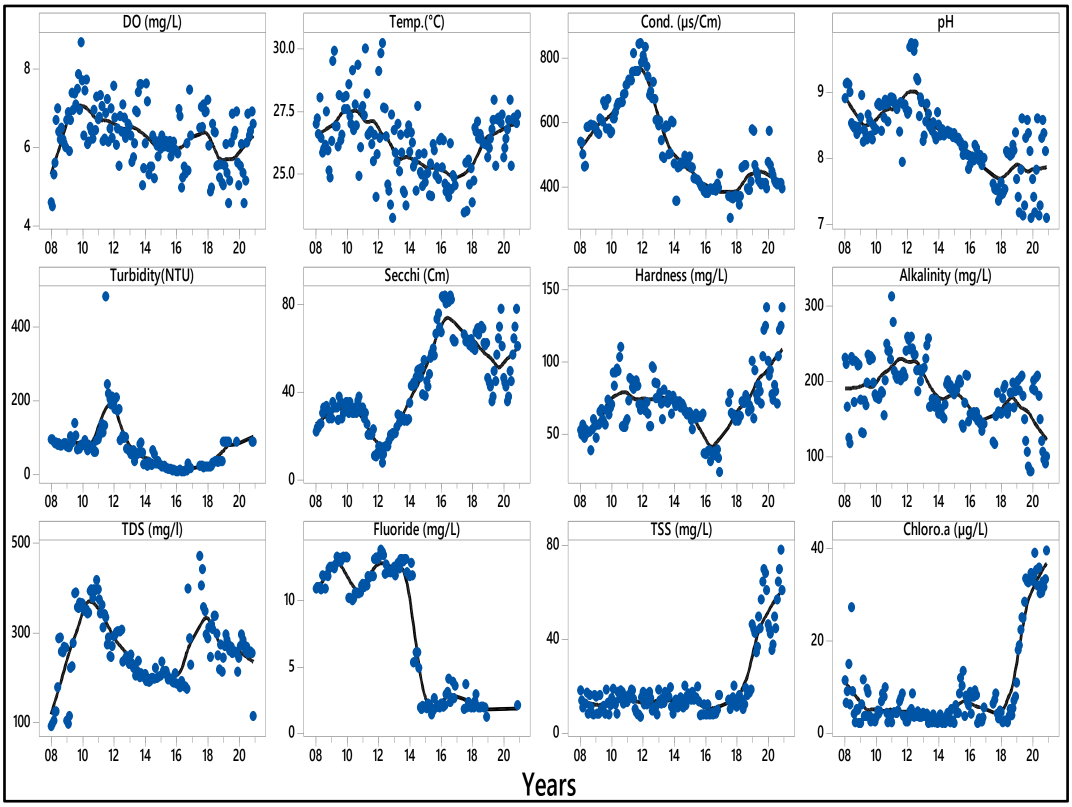

Figure 4.1: Temporal variation of physico-chemical water quality variables in Lake Baringo, Kenya, for the period 2008-2020. Circles are means of all samples collected monthly from stations while the smoothing trendline is estimated by LOWESS smoother.

Nutrient concentrations (TN, TP, PO_4^{3-}, SiO_4^{4-}, NO_2^-, NO_3^- and NH_4^+) showed trends that varied between parameters from 2008 to 2020 (Figure 4.2). Nitrites (NO_2^-) increased in the lake by 94.7% from 2008 (68.2 µg L^{-1}) to 2014 (603.0 µg L^{-1}) and then decreased continuously up to ~ 24.3 µg L^{-1} in 2020. Nitrates (NO_3^-) mean levels have fluctuated from 5.3 µg L^{-1} in 2008 to 14.7 µg L^{-1} in 2013 while, NH_4^+ levels ranged between 23.62 µg L^{-1} in 2008 to a peak of 42.0 µg L^{-1} in 2016. Phosphates (PO_4^{3-}) that reflect leaching from the riparian zone peaked in 2011 (34 µg L^{-1}) and have remained fairly stable in the lake at about 5.6 µg L^{-1} between 2013 and 2020.

This contrasts with Silicates (SiO_4^{4-}) that have shown a general decline in the lake from high values in 2008 compared to other nutrient components. Total phosphorous (TP) has shown a steady decline from peak values in 2011 (180.9 μg L^{-1}) wish is similar to the trends of Total Nitrogen (TN) in the lake.

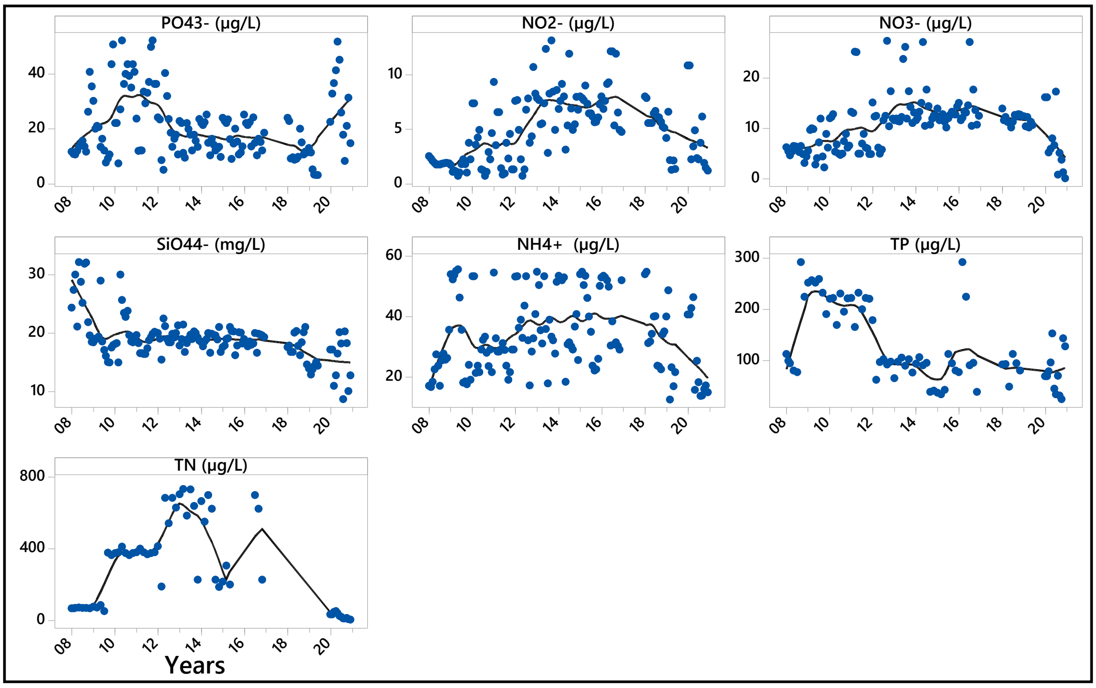

Figure 4.2: Time-series of nutrient concentrations in Lake Baringo, Kenya. Circles are means of all samples collected monthly in the sampling stations from 2008 to 2020. The solid line within the time-series plot is the trendline of monthly means calculated using LOWESS smoother.

The short-term (intra-annual) data analysis (2020-2021) indicated that the mean (± SD) of six parameters (DO, pH, Secchi disc, TDS, NO_2^-, and SiO_4^{4-}) were significantly different ($p < 0.05$) between the nine stations in the Lake (Table 4.2). Five variables (DO, Chl-a, TP, NO_2^-, and SiO_4^{4-}) including the water quality index (WQI) were above the threshold values recommended for both human consumption and aquatic life. Other parameters (temperature, conductivity, pH, alkalinity, hardness, alkalinity, TN, PO_4^{3-}, NO_3^-, NH_4^+ and organic pollution index, OPI) did not show significant differences ($p > 0.05$) between stations and were within the standard limits recommended for human consumption and maintenance of ecological processes (Table 4.2).

Dissolved oxygen (DO) was marginally high in the central station C1 (7.08 ± 1.56 mg L^{-1}) and lowest in the southern station S2 (4.87 ± 1.39 mg L^{-1}) while, pH showed higher mean values in the northern station N3 (7.89 ± 0.62) and lowest values were found in the southern station S2 (7.43 ± 0.39 °C). Secchi disc measurements were significantly different between the stations with the highest mean values in the northern station N2 (66.56 ± 18.30 cm) and lowest in the near-shoreline station S3 (50.37 ± 15.47 cm). TDS mean levels also showed significant differences between the stations with peak values in the central station C1 (289.09 ± 18.98 mg L^{-1}) and lowest in the southern near-shoreline station S2 (235.36 ± 71.17 mg L^{-1}).

NO_2^- was significantly different between the stations with higher values in the central lake station C1 (7.36 ± 10.97 µg L^{-1}) and lowest in the southern near-shoreline station S2 (1.51 ± 1.25 µg L^{-1}). Silicate ions (SiO_4^{4-}) levels were significantly different between the stations with higher values in the central lake station C2 (20.22 ± 8.06

mg L^{-1}) and lowest in the southern near-shoreline station C1 (9.84 ± 6.92 mg L^{-1}). Both $NO_2^{\ -}$ and SiO_4^{4-} ion contents were above the WHO recommended levels (3000 µg L^{-1} and 5 mg L^{-1}) for aquatic conditions.

Table 4.2: Statistical summary of some physico-chemical parameters of Lake Baringo, Kenya, for the period January 2020 – June 2021, Kenya. Bold figures are significantly different between stations or were above value criteria (WHO/APHA thresholds and/or others, Table 4.1).

	Sampling sites										
	S1	S2	S3	C1	C2	C3	N1	N2	N3	ANOVA	
Parameters	Mean±SD	Mean±SD	Mean±SD	Mean±SD	Mean±SD	Mean±SD	Mean±SD	Mean±SD	Mean±SD	F	P
DO (mg/L)	**5.74±0.74**	**4.87±1.39**	**6.29±1.56**	**7.08±2.33**	**6.64±1.63**	**6.58±1.61**	**6.45±1.27**	**6.68±1.79**	**6.55±1.27**	**3.09**	**0.003**
Temp. (°C)	26.65±0.79	27.18±1.11	26.82±0.80	27,04±1.44	27.15±1.26	27.06±1.02	27.73±1.38	27.10±1.15	25.70±4.10	1.80	0.081
EC (µs/cm)	459.75±70.34	417.09±103.03	454.68±72.31	469.70±68.53	464.44±70.70	463.28±66.75	475.44±64.92	469.54±67.02	413.84±111.65	1.50	0.163
pH	**7.57±0.55**	**7.43±0.39**	**7.55±0.61**	**7.60±0.71**	**7.82±0.77**	**7.74±0.61**	**7.69±0.64**	**7.72±0.62**	**7.89±0.62**	**2.03**	**0.046**
SD (cm)	**53.28±14.63**	**50.96±22.71**	**50.37±15.47**	**63.03±20.03**	**64.05±18.61**	**57.75±16.85**	**65.11±18.41**	**66.56±18.30**	**62.17±19.16**	**2.17**	**0.033**
HD (mg/L)	71.25±23.82	78.24±53.94	89.41±68.66	59.62±39.72	75.56±40.86	71.73±37.19	44.58±41.16	48.47±40.43	66.43±32.16	1.82	0.078
Alk (mg/L)	139.28±50.90	120.23±51.86	118.80±33.48	119.38±30.97	110.65±41.99	129.73±43.56	117.64±60.40	93.51±25.78	101.02±54.60	1.38	0.214
TDS (mg/L)	**268.48±24.62**	**235.36±71.17**	**277.72±23.72**	**289.09±18.98**	**288.09±20.13**	**284.48±20.04**	**288.23±19.36**	**282.63±20.56**	**258.94±74.03**	**3.80**	**0.000**
PO_4^{3-} (µg/L)	12.29±15.11	8.13±9.82	11.56±14.42	15.31±26.77	5.72±12.46	10.79±14.45	15.70±22.87	299.25±53.14	13.50±34.30	1.18	0.313
TP (µg/L)	**72.43±49.49**	**51.13±49.69**	**66.17±68.71**	**35.10±41.39**	**32.21±46.77**	**47.91±64.55**	**38.86±41.02**	**23.43±31.41**	**54.91±80.52**	1.57	0.139
NO_2^- (µg/L)	**6.67±9.15**	**1.51±1.25**	**5.32±6.11**	**7.36±10.97**	**2.33±1.63**	**3.98±5.57**	**4.89±3.90**	**6.93±8.83**	**1.96±1.48**	**2.18**	**0.032**
NO_3^- (µg/L)	9.59±10.08	5.64±10.40	8.73±8.05	5.66±7.46	6.15±3.04	6.63±6.43	7.23±5.38	5.91±3.92	25.69±63.63	1.48	0.168
TN (µg/L)	24.16±17.66	33.68±26.63	31.22±28.50	52.39±51.06	24.57±14.27	30.85±26.59	37.41±30.08	48.83±60.36	21.26±11.84	1.86	0.066
SiO_4^{4-} (mg/L)	**16.15±6.57**	**15.59±10.78**	**17.09±8.19**	**9.84±6.92**	**20.22±8.06**	**13.96±7.82**	**11.36±7.74**	**15.24±13.67**	**12.50±8.88**	**2.22**	**0.029**
Chl-a (µg/L)	**57.82±72.06**	**24.55±26.76**	**43.84±60.74**	**51.64±69.34**	**23.36±24.19**	**59.62±66.29**	**63.05±64.09**	**52.63±67.00**	**22.78±30.21**	1.53	0.151
NH_4^+(mg/L)	24.82±43.71	19.17±22.16	21.82±24.95	31.50±53.78	16.56±10.90	12.19±5.91	37.84±58.52	45.26±53.13	27.90±36.96	1.32	0.239
WQI	**401.45±95.54**	**405.64±118.90**	**427.37±110.89**	**450.61±152.66**	**342.93±125.93**	**440.83±45.52**	**441.13±43.76**	**445.92±98.88**	**442.27±101.17**	1.95	0.057
OPI	4.65±0.27	4.65±0.29	4.57±0.23	4.56±0.23	4.52±0.31	4.52±0.31	4.63±0.30	4.54±0.26	4.58±0.28	0.64	0.739

4.1.2 Relationship between limnological parameters in the lake

Linear models showed a moderate negative relationship between chlorophyll-a as a measure of productivity and total suspended solids ($R^2 = 0.59$) (Figure 4.3a) while, a strong positive linear relation was found between turbidity and rainfall in the lake catchment ($R^2 = 0.71$) (Figure 4.3b).

Pearson's correlation matrix was generated using 21 variables in order to determine the functional relationships between the limnological variables (Table 4.3). WQI showed a positive significant correlation only with turbidity ($p < 0.001$, r = 0.999) (Table 4.3). There was a positive significant correlation between alkalinity and temperature ($p = 0.004$, r = 0.978). Positive significant relationships were also obtained between DO and Secchi depth ($p = 0.026$, r = 0.922) and between DO and alkalinity ($p = 0.024$, r = 0.926). A strong significant correlation was observed between $S_iO_4^{4-}$ and alkalinity ($p = 0.02$, r = 0.933) and between chlorophyll-a, and NO_3^- ($p = 0.039$, r = 0. 898). Negative relationships were obtained between $S_iO_4^{4-}$ and total hardness ($p = 0.033$, r = – 0.908) and between total suspended solid (TSS) and total nitrogen (TN) ($p = 0.013$, r = – 0.950).

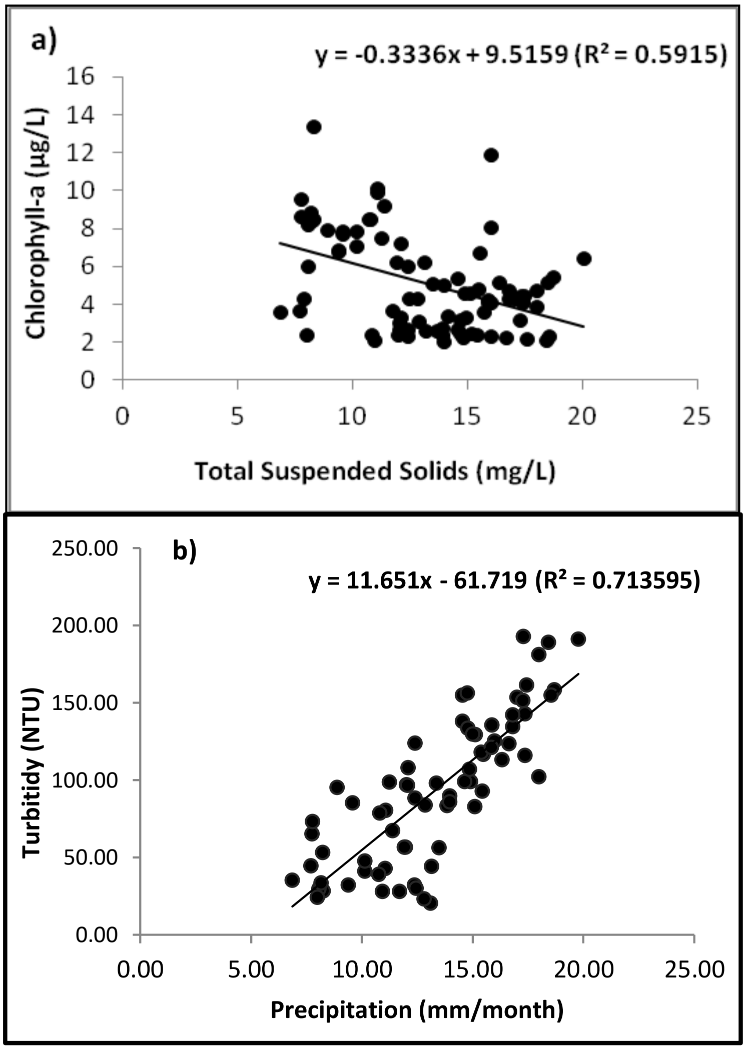

Figure 4.3: Relationship between a) Chlorophyll-a and total suspended solids, b) turbidity of lake water and rainfall in the catchment of Lake Baringo, Kenya, for the period 2008-2020.

Table 4.3: Pearson's linear correlation matrix of physico-chemical parameters derived within Lake Baringo, Kenya, from March 2008 to June 2021.

	DO	Temp	EC	pH	TUR	Secchi	HD	Alk.	Depth	TDS	F^-	PO_4^{3-}	TP	NO_2^-	NO_3^-	TN	$S_iO_4^{4-}$	NH_4^+	Chl.a	TSS	WQI
DO	1.00																				
Temp	0.92*	1.00																			
EC	0.73	0.44	1.00																		
pH	0.82	0.74	0.77	1.00																	
TUR	-0.15	-0.23	0.29	0.44	1.00																
Secchi	0.92*	0.77	0.77	0.63	-0.31	1.00															
HD	-0.72	-0.88	-0.26	-0.57	0.25	-0.66	1.00														
Alk.	0.93*	0.98**	0.51	0.76	-0.20	0.83	-0.93*	1.00													
Depth	0.62	0.45	0.69	0.40	-0.21	0.84	-0.58	0.61	1.00												
TDS	-0.62	-0.71	-0.15	-0.53	0.13	-0.32	0.34	-0.56	0.22	1.00											
F^-	0.13	0.01	0.21	0.30	0.35	-0.12	0.45	-0.14	-0.53	-0.62	1.00										
PO_4^{3-}	-0.57	-0.32	-0.54	-0.16	0.51	-0.69	0.01	-0.30	-0.41	0.35	-0.30	1.00									
TP	0.53	0.18	0.87	0.37	-0.04	0.75	-0.09	0.30	0.80	0.16	-0.04	-0.71	1.00								
NO_2^-	0.40	0.71	-0.18	0.39	-0.11	0.19	-0.84	0.67	0.07	-0.45	-0.30	0.38	-0.45	1.00							
NO_3^-	0.30	0.65	-0.31	0.31	-0.10	0.04	-0.71	0.58	-0.15	-0.55	-0.14	0.40	-0.61	0.97**	1.00						
TN	0.60	0.45	0.48	0.53	0.01	0.40	0.03	0.33	-0.11	-0.82	0.85	-0.66	0.27	-0.10	-0.01	1.00					
$S_iO_4^{4-}$	0.85	0.84	0.60	0.67	-0.21	0.89*	-0.91*	0.93*	0.84	-0.23	-0.40	-0.30	0.50	0.55	0.37	0.09	1.00				
NH_4^+	-0.22	-0.46	0.40	-0.03	0.41	0.05	0.21	-0.27	0.50	0.84	-0.40	0.12	0.56	-0.50	-0.66	-0.50	0.05	1.00			
Chl.a	0.36	0.64	-0.21	0.41	-0.00	0.02	-0.49	0.51	-0.37	-0.83	0.31	0.21	-0.58	0.80	0.90*	0.38	0.19	-0.84	1.00		
TSS	-0.61	-0.40	-0.68	-0.66	-0.25	-0.44	-0.06	-0.31	-0.03	0.67	-0.83	0.60	-0.44	0.20	0.15	-0.95*	-0.14	0.23	-0.23	1.00	
WQI	-0.11	-0.20	0.32	0.47	0.99***	-0.28	0.24	-0.18	-0.20	0.09	0.38	0.48	-0.03	-0.11	-0.10	0.06	-0.19	0.38	0.02	-0.30	1.00

$P<0.001$*** $P<0.01$** $P<0.05$*

4.1.3 Water quality (WQI) and Organic Pollution (OPI) Indices

Water quality index (WQI) values of Lake Baringo water samples varied between 540.85 at Station N2 on the north to its lowest value of 631.89 obtained at central Station C3. There was no significant difference in the WQI between the sampling sites (F = 0.6816; p = 0.6077). The mean monthly WQI exhibited similar patterns of variation to those of the mean monthly turbidity level in the lake for the period from January 2008 to May 2019 (Figure 4.4), highlighting the influence of turbidity levels on the lake's WQI. Both highest and lowest WQI values peak above the threshold of 100 recommended by WHO and APHA for both human and aquatic conditions. The highest WQI values were observed during the rainy season (May 2011), while the lowest WQI values were observed during the dry month of September 2016 at ~120, indicating a relatively good water quality during the dry months (Figure 4.4).

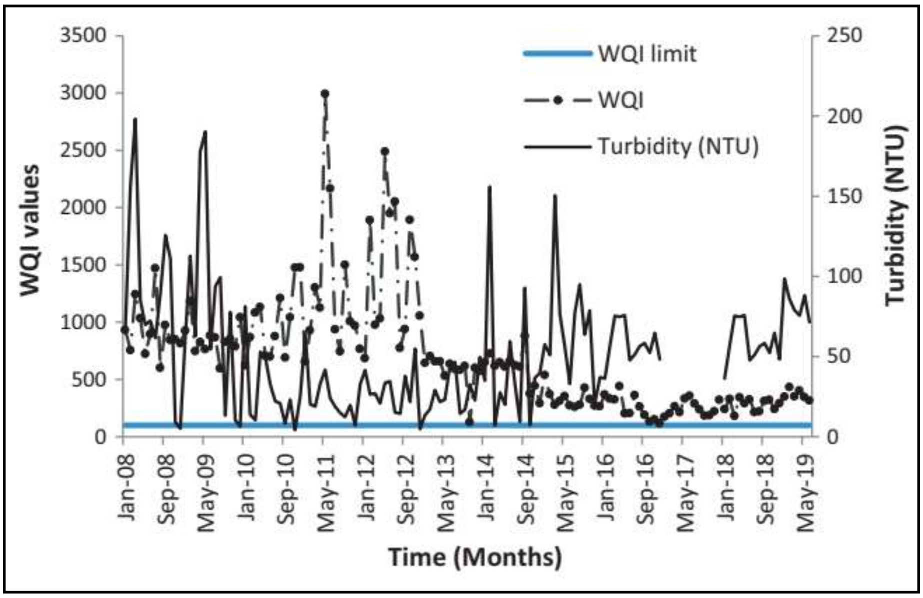

Figure 4.4: Changes in the monthly water quality index (WQI) in relation to turbidity in Lake Baringo, Kenya, for the period 2008-2020. Blue line indicates upper limit of WQI as per WHO standards.

The relative (RW) and effective (EW) weight values of each water quality variable are summarized in Table 4.4. The turbidity exhibited the highest mean effective weight of 84.82%, followed by fluoride (8.82%), indicating that the two parameters influence mainly Lake Baringo's water quality evaluated using the WQI. The other variables (mostly nutrients) exhibited a low effective influence on the lake's WQI. Nevertheless, the reactive soluble silica (SiO_4^{4-}) and pH had a moderately affect the quality of the water in the lake with effective weight of 3.09% and 1.50% respectively.

Table 4.4: The effective weight contribution of physico-chemical parameters to Water Quality Index (WQI) in Lake Baringo, Kenya, in relation to the WHO standards during the period 2008-2020.

Parameters	WHO (2005, 2008)	Assigned weight (wi)	Relative weight (RW)	**Effective weight (%)**			
				Min	Max	Mean	SD
DO (mg/L)	5.00	4.00	4.65	0.86	1.08	0.95	0.11
EC (µS/cm)	1000.00	1.00	2.33	0.18	0.22	0.20	0.02
pH	8.50	4.00	9.30	1.33	1.66	1.50	0.13
TUR (NTU)	5.00	4.00	9.30	83.24	86.42	84.82	1.33
HD (mg/L)	500.00	2.00	4.65	0.08	0.10	0.09	0.01
Alk. (mg/L)	500.00	1.00	2.33	0.12	0.15	0.14	0.01
TDS (mg/L)	500.00	2.00	4.65	0.31	0.38	0.35	0.03
F^- (mg/L)	1.50	5.00	11.63	7.76	9.51	8.82	0.73
PO_4^{3-} (mg/L)	0.03	5.00	11.63	0.00	0.00	0.00	0.00
NO_2^- (mg/L)	0.30	5.00	11.63	0.00	0.01	0.00	0.00
NO_3^- (mg/L)	1.00	5.00	11.63	0.00	0.00	0.00	0.00
SiO_4^{4-} (mg/L)	5.00	2.00	4.65	2.63	3.61	3.09	0.42
NH_4^+ (mg/L)	1.50	5.00	11.63	0.03	0.04	0.04	0.01

The Organic pollution index (OPI) values varied between 4.5 at Station C2 to 4.9 at Station C3, with a significant difference noted among the sampling sites (F = 3.59, p = .013; Table 4.5). These results showed that the OPI of Lake Baringo's water is within the threshold limits of water exhibiting null organic pollution (5.0–4.6; Table 3.4), except for one central station C2 characterized by weak organic pollution manifesting in a greenish water color.

Table 4.5: Organic pollution index (OPI) values and organic pollution types of sampling sites in Lake Baringo during the period 2008-2020. Different letters indicate significant differences between stations.

Sample ID	OPI (mean ±SD)	Organic pollution	Colors of index
S2	$4.6 \pm 0.19^{\mathbf{ab}}$	Null	Blue
C1	$4.6 \pm 0.29^{\mathbf{ab}}$	Null	Blue
C2	$4.5 \pm 0.34^{\mathbf{b}}$	Weak	Green
C3	$4.7 \pm 0.29^{\mathbf{ab}}$	Null	Blue
N2	$4.9 \pm 0.14^{\mathbf{a}}$	Null	Blue
ANOVA $F = 3.59$, $p = 0.013$			

4.1.4 Trophic status of the lake

The results of TN:TP and DIN:SRP ratios and of Trophic Status Index (TSI) analyses are shown in Tables 4.6 and 4.7 for inter- and intra-annual patterns, respectively. The mean (±SD) stoichiometric ratio TN:TP varied from the highest value of 6.91 ± 2.66 in 2013 to the lowest value of 0.38 ± 0.21 in 2020, while the mean (± SD) seston mass ratio DIN:SRP fluctuated overtime between 4.42 ± 1.37 in 2018 and 1.45 ± 0.84 in 2020. These TN:TP and DIN: SRP ratios indicate that the lake is eutrophic and that the nitrogen component (NO_2^-) is the likely limiting nutrient for primary production in Lake Baringo.

The mean (± SD) Trophic State Index (TSI) calculated based on total phosphorus (TP) contents in the lake waters varied from a high value of 84.24 ± 0.52 in 2009 to its lowest value of 21.92 ± 4.90 in 2014. The TSI determined based on Secchi disk (SD) depth also varied from a high level of 88.00 ± 2.99 in 2012 to the lowest level of 55.63 ± 0.59 in 2019. While, the TSI based on the lake productivity (Chl-a values) varied from a high value of 83.50 ± 0.78 in 2009 to its lowest value of 42.69 ± 3.33 in 2014 (Table 4.6). The Carlson's Trophic State Index (CTSI) that integrates all the three values varied from the highest value of 82.68 ± 2.10 in 2011 indicating a Hypereutrophic state of the lake to its lowest value of 42.61 ± 2.49 indicating a Mesotrophic state of the lake in 2015 (Table 4.6). All the CSTI values indicated that Lake Baringo's trophic status has been fluctuating from Hypereutrophic to Mesotrophic state and from water of bad quality to water of moderate quality for domestic supply. The current (2020) trophic state of Lake Baringo is derived as being Eutrophic (Table 4.6). The results show that CTSI is mostly affected by the suspended solid (SD) values in Lake Baringo.

Table 4.6: Temporal variation of TN:TP and DIN:SRP ratios as indicators of phytoplankton nutrient limitation in the lake, and Trophic Status Indices in Lake Baringo, Kenya, for the period 2008-2020.

Years				Trophic State Index (TSI) values				
	n	TN:TP	DIN:SRP	TSI (TP)	TSI (SD)	TSI (Chla)	CTSI	Lake status
2008	11	0.62 ± 0.27	2.04 ± 0.69	31.51 ± 9.36	78.50 ± 2.30	49.22 ± 6.37	53.07 ± 5.31	mesotrophic
2009	12	0.77 ± 0.78	2.75 ± 1.21	84.24 ± 0.52	76.47 ± 1.60	83.50 ± 0.78	81.30 ± 0.79	hypereutrophic
2010	11	1.84 ± 0.30	1.65 ± 1.42	83.02 ± 4.60	76.09 ± 1.10	81.2 ± 1.18	80.14 ± 0.78	hypereutrophic
2011	11	1.84 ± 0.29	2.08 ± 1.74	83.13 ± 0.58	83.57 ± 5.02	81.37 ± 0.84	82.68 ± 2.10	hypereutrophic
2012	12	5.27 ± 2.15	3.82 ± 2.29	76.17 ± 2.93	88.00 ± 2.99	71.13 ± 3.52	78.43 ± 2.01	eutrophic
2013	12	6.91 ± 2.66	3.68 ± 0.78	74.64 ± 1.07	79.34 ± 1.28	68.88 ± 1.56	74.29 ± 0.73	eutrophic
2014	12	6.19 ± 1.19	3.78 ± 0.57	21.92 ± 4.90	71.16 ± 2.05	42.69 ± 3.33	45.27 ± 3.10	mesotrophic
2015	11	3.3 ± 3.90	3.73 ± 1.13	25.98 ± 5.54	56.40 ± 1.91	45.45 ± 3.77	42.61 ± 3.62	mesotrophic
2016	11	3.37 ± 3.74	3.76 ± 0.88	28.38 ± 6.71	61.34 ± 1.93	47.08 ± 4.56	45.60 ± 3.69	mesotrophic
2017	6	3.34 ± 3.82	4.09 ± 1.13	26.80 ± 6.11	62.86 ± 3.10	46.01 ± 4.16	45.22 ± 3.98	mesotrophic
2018	11	3.36 ± 3.78	4.42 ± 1.37	25.22 ± 5.51	64.37 ± 4.27	44.94 ± 3.75	44.84 ± 4.08	mesotrophic
2019	6	1.87 ± 2.01	2.94 ± 1.12	27.42 ± 4.18	55.63 ± 0.59	46.13 ± 2.85	43.17 ± 2.49	mesotrophic
2020	12	0.38 ± 0.21	1.45 ± 0.84	55.10 ± 10.49	70.90 ± 1.16	65.26 ± 7.44	63.76 ± 6.14	eutrophic

n: number of monthly samples, TN: total nitrogen, TP = total phosphorous, DIN : dissolved inorganic nitrogen, SRP : soluble reactive phosphorus, TSI (TP): Trophic Status Index based on total phosphorus, TSI (SD): Trophic Status Index based on secchi disk, TSI (Chl-a): Trophic Status Index based on chlorophyll-a and CTSI stands for Carlson's Trophic Status Index.

At an intra-annual scale, based on 2020-2021 data, the stoichiometric ratio TN: TP varied from the highest mean (± SD) value of 11.35 ± 4.91 in April 2021 to the lowest value of 0.03 ± 0.001 in December 2020 while, the seston mass DIN:SRP fluctuated overtime between 236.70 ± 10.94 in May 2021 and 0.55 ± 0.06 in May 2020 (Table 4.7). These TN:TP and DIN: SRP ratios indicate that the lake is fluctuating monthly from being Eutrophic to Oligotrophic (also shown by long-term data) and that the nitrogen component (NO_2^-) is still the limiting nutrient for primary production into the lake. The monthly mean (± SD) TSI calculated based on total phosphorus (TP) contents in the lake waters varied from a high value of 66.64 ± 2.65 in February 2020 to its lowest value of 9.52 ± 44.63 in June 2021. The TSI determined based on monthly Secchi disk (SD) depth also varied from a high level of 72.99 ± 1.12 in May 2020 to the lowest level of 61.94 ± 0.23 in January 2021. The TSI based on the lake productivity (Chl-a values) varied from a high value of 73.11 ± 10.20 in February 2020 to its lowest value of 25.49 ± 6.23 in January 2021 (Table 4.7).

The Carlson's Trophic State Index (CTSI) varied from the highest value of 70.34 ± 2.61 in February 2020 indicating a Eutrophic state of the lake to its lowest value of 33.08 ± 0.27 indicating an Oligotrophic state of the lake in June 2021. All the CSTI values indicated that Lake Baringo's trophic status has been fluctuating monthly from Eutrophic to Oligotrophic state, and from water of bad quality to water of moderate quality for domestic supply. The most current (from January 2021) trophic state of Lake Baringo is derived as being Mesotrophic. The results show that the monthly CTSI is also mostly affected by the suspended solid (SD) values as also shown by the annual CTSI values in Lake Baringo (Table 4.7).

Table 4.7: Monthly variation of TN:TP and DIN:SRP ratios as nutrient limitation indicators and Trophic Status Indices of Lake Baringo from January 2020 to June 2021.

Months	TN:TP	DIN:SRP	TSI (TP)	TSI (SD)	TSI (Chla)	CTSI	Lake status
Jan. 20	0.48 ± 0.02	3.02 ± 0.16	49.62±3.65	71.13±1.09	61.54±8.46	60.76±3.42	eutrophic
Feb. 20	0.48 ± 0.05	2.07 ± 0.91	66.64±2.65	71.25±0.98	73.11±10.20	70.34±2.61	eutrophic
Mar. 20	0.60 ± 0.03	1.37 ± 0.16	55.27±7.15	71.25±0.87	65.38±7.21	63.97±3.98	eutrophic
Apr. 20	0.53 ± 0.09	1.36 ± 0.18	61.93±6.49	72.69±1.02	69.91±8.45	68.17±3.12	eutrophic
May 20	0.26 ± 0.01	0.55 ± 0.06	63.77±9.31	72.99±1.12	71.16±10.68	69.30±2.76	eutrophic
June 20	0.52 ± 0.07	0.76 ± 0.05	31.16±7.48	72.84±0.89	48.97±6.45	50.99±2.98	mesotrophic
July 20	0.61 ± 0.05	1.46 ± 0.12	54.02±3.54	71.24±0.53	64.53±9.65	63.26±4.85	eutrophic
Aug. 20	0.17 ± 0.02	1.02 ± 0.41	58.83±9.53	70.48±0.43	67.80±12.03	65.71±1.69	eutrophic
Sept. 20	0.35 ± 0.01	3.02 ± 1.03	51.49±11.43	70.61±1.01	62.80±7.89	61.63±2.93	eutrophic
Oct. 20	0.53 ± 0.11	1.03 ± 0.62	41.44±2.56	68.46±0.88	55.97±5.43	55.29±1.26	mesotrophic
Nov. 20	0.04±0.002	0.64 ± 0.08	64.25±4.91	68.98±0.13	71.49±7.31	68.24±4.19	eutrophic
Dec. 20	0.03±0.001	1.10 ± 0.17	62.75±8.56	68.80±0.06	70.46±10.82	67.34±3.56	eutrophic
Jan. 21	0.44 ± 0.01	167.36±15.64	54.99±10.3	61.94±0.23	44.84±8.42	53.92±2.89	eutrophic
Feb. 21	1.61 ± 0.09	277.10±12.93	31.61 ± 11.4	62.89±0.65	61.94±10.34	52.15±1.09	eutrophic
Mar. 21	5.21 ± 1.08	280.07±21.17	27.95±12.14	64.66±1.01	34.56±4.81	42.39±0.93	mesotrophic
Apr. 21	11.35±4.91	175.73±11.78	21.22±9.65	66.60±0.21	47.12±6.86	44.98±1.07	mesotrophic
May 21	6.05 ± 2.07	236.70±10.94	15.65±13.3	63.60±0.12	40,74±8.21	39.99±0.85	oligotrophic
June 21	7.98 ± 3.15	205.68±11.25	9.52±44.63	64.24±1.23	25.49±6.23	33.08±0.27	oligotrophic

4.2. Objective 2: Water balance modeling and its sensitivity to hydro-meteorological variables

4.2.1 Water level and rainfall trends

The annual rainfall in Lake Baringo basin ranged from 225 mm yr^{-1} in 2009 to 2382 mm yr^{-1} in 2019 (Table 4.8) with the annual mean (± SD) rainfall estimated at 959 ± 361 mm yr^{-1} for the period from 1960 to 2020, while the annual lake level varied from 0.9 m in 2001 to 6.18 m in 2020 giving an annual mean of 4.25 ± 1.25 m for the long-term period 1960-2020 (Figure 4.5).

At short temperal scale (January 2020-June 2021), the monthly lake level (Figure 4.5b) varied from 11.04 m in January 2020 to a peak of 14.52 m in November 2020 with a monthly mean (± SD) estimated at 13.19 ± 1.29 m for the period from January 2020 to June 2021. The monthly lake level increased dramatically during rainy season between April 2020 (11.31 m) and November 2020 (14.52 m). It started decreasing progressively from 14.28 m in December 2020 to 13.90 m in June 2021 (Figure 4.5b) indicating the importance of the rainfall at short-term scale in the lake level changes. This is highlighted in the relationship between the precipitation in the lake basin and the lake level at long-term scale (1960-2020) (Figure 4.5a).

Table 4.8: (a) Monthly mean (± SD) and (b) total annual evaporation and rainfall values over Lake Baringo's catchment as estimated by Thornthwaite method for the period from 2008 to 2021. Monthly means are compared over the years.

	Month												
(a) Meteo. variables	Jan.	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.	Total
Evaporation (mm)	147±6.3	166±8.4	153±8.3	151±8.8	139±8.4	138±6.0	132±6.3	136±7.6	143±7.6	142±6.2	145±8.9	142±8.1	**1734**
Precipitation (mm)	16 ±63.8	18 ±19.1	53±53.0	116±155.4	109±137.3	80±90.0	77±94.8	132±149.0	99±112.3	103±117.1	83±75.2	62±79.9	**943**

	Years													
(b) Meteo. variables	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	2021
Evaporation (mm)	1741	1781	1714	1678	1667	1697	1696	1720	1699	1715	1732	1880	1805	1550
Precipitation (mm)	629	225	1033	1177	792	852	578	888	833	700	962	2382	3707	2166

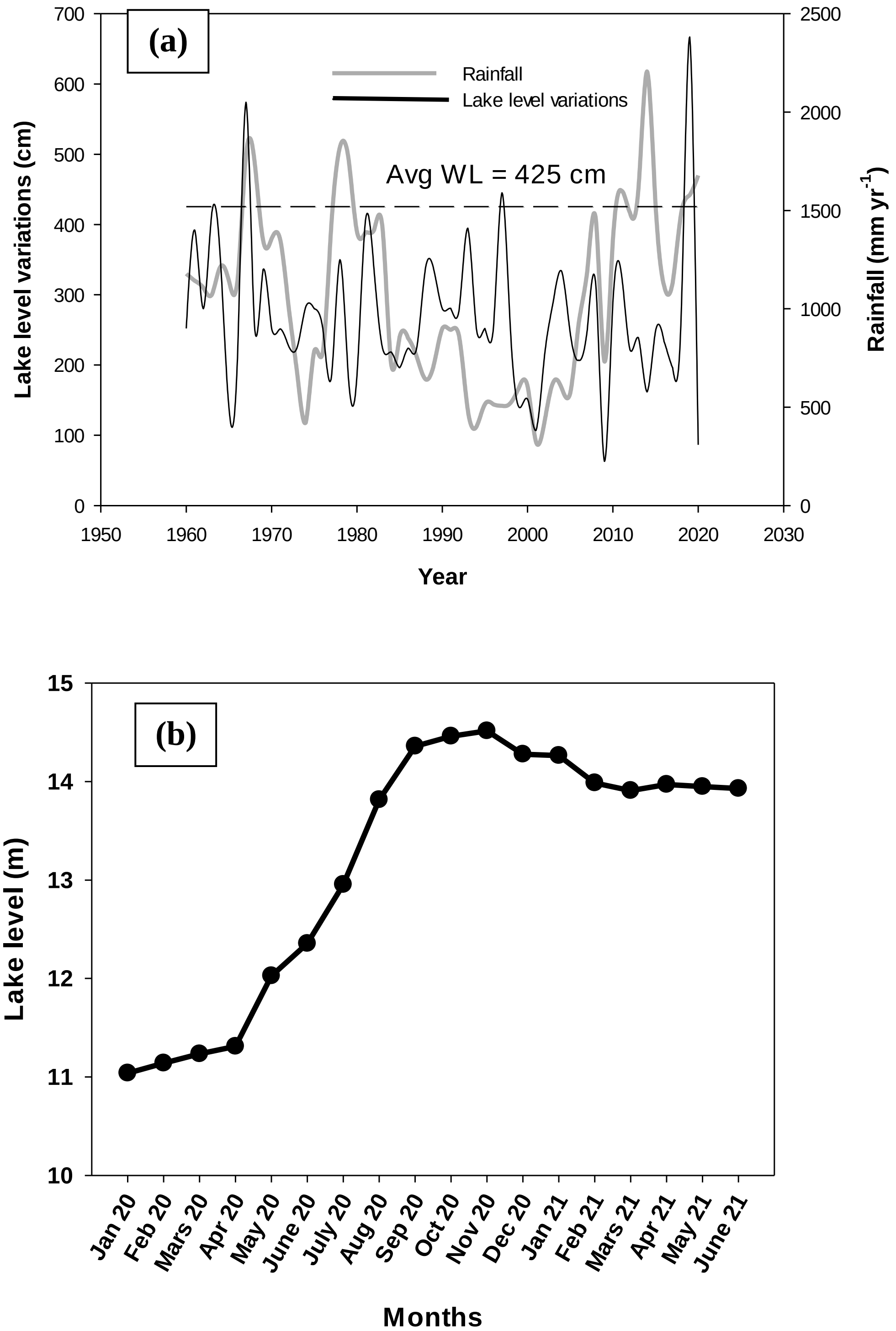

Figure 4.5: (a) Variations of annual Lake Baringo water level (black continuous line) and precipitation (grey line) for the period 1960-2020 with average long-term lake level (dashed line) and (b) monthly lake levels for the period January 2020-June 2021.

The depth-lake storage capacity curve for the lake was found to be linear ($r^2 = 0.965$, $p = 0.0001$) and highlighted the dependence of the changes in the lake level (volume) on the precipitation on the lake surface (Figure 4.6). In current conditions, the findings indicate that the lake can store upto 3167 Km^3 of rainwater (surface runoff and directly falling into the lake) at the maximum storage depth of 14.4 m (depth as at November 2020) (Figure 4.6).

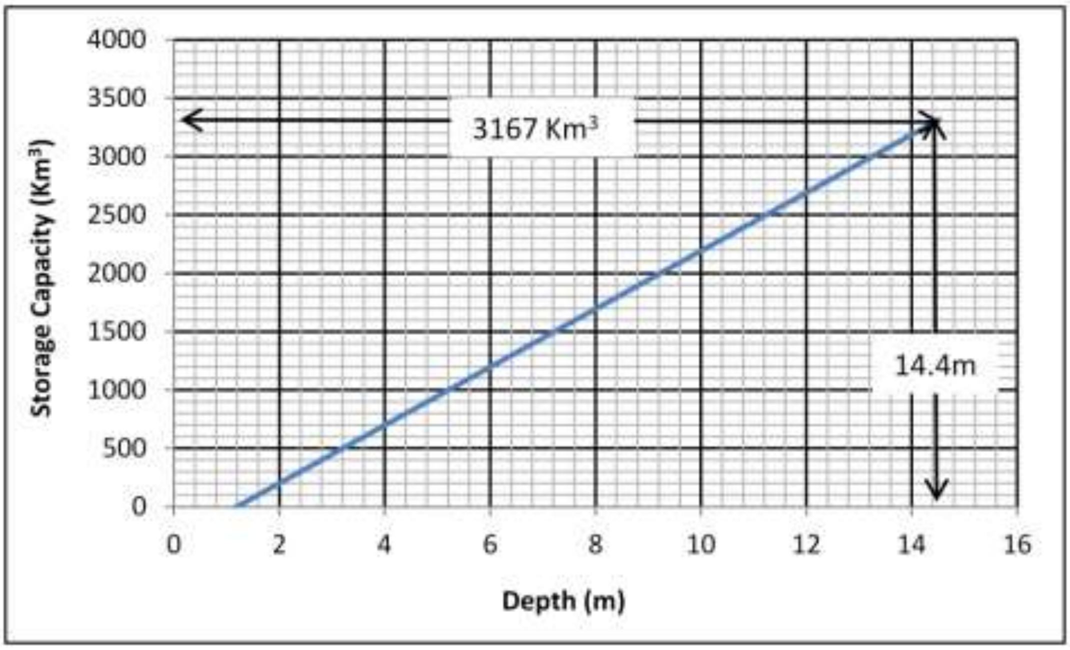

Figure 4.6: The depth – capacity curve for Lake Baringo, Kenya, for the period January 1970 - June 2021.

An established comparison between lake levels observed *in-situ* and remote sensed lake levels for some neighboring Rift Valley Lakes such as Lakes Turkana and Naivasha showed very similar temporal dynamics as for Lake Baringo (Figure 4.7). These trends indicate that the meteorological variables in the region are the main drivers of the lake level fluctuations in the Rift Valley Lakes.

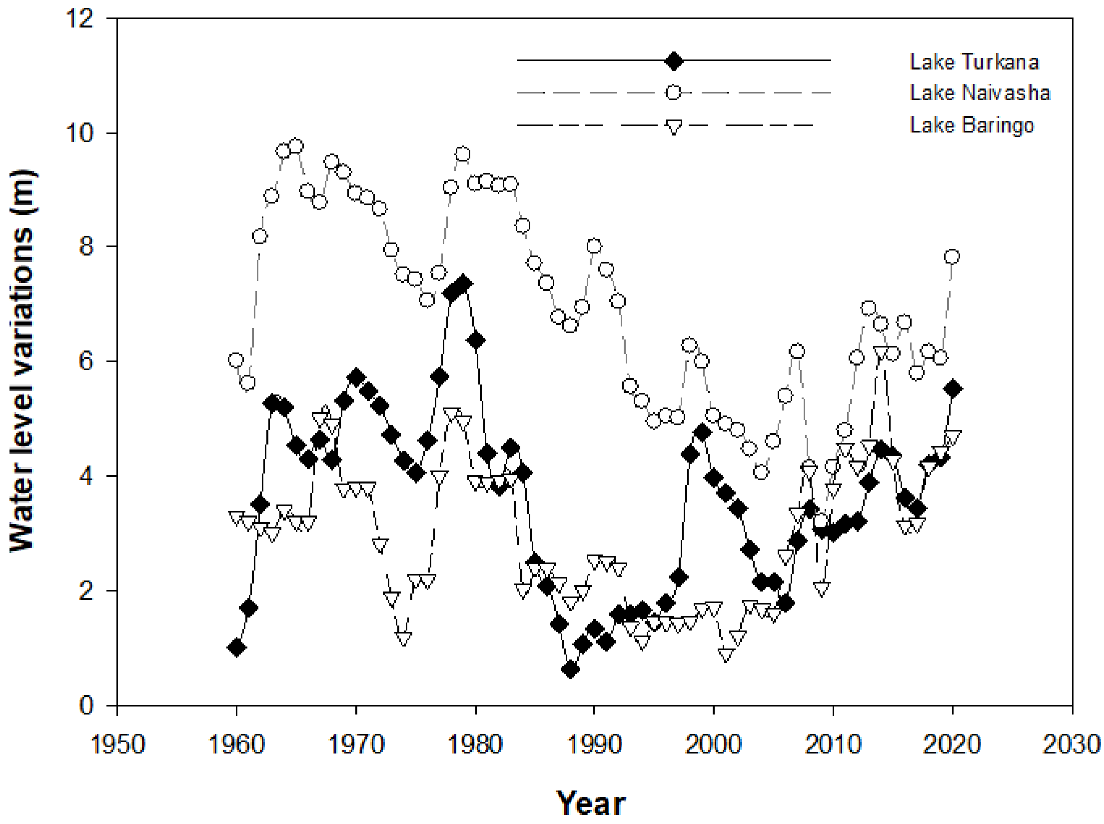

Figure 4.7: Comparison of water level variations (relative average levels) of selected neighboring rift valley lakes with the levels observed from in-situ gauged data for Lake Baringo, Kenya (period 1960-2020). Data from Lakes Turkana and Naivasha are satellite derived.

4.2.2 Evaporation and rainfall in Lake Baringo basin

The amount of water lost from the lake surface in the form of evaporation (E) at intra-annual scale varied from 132 ± 6.3 mm month^{-1} in July to 166 ± 8.4 mm month^{-1} in February for the period 2008-2021 (Table 4.8a). At annual scale, evaporation varied 1550 mm yr^{-1} in 2021 to 1880 mm yr^{-1} in 2019 for the period 2008-2021 (Table 4.8b).

At mothly scale, the lake evaporation was higher from January to April (147 ± 6.8 mm month^{-1} to 166 ± 8.4 mm month^{-1}) with a peak measured in February (166 ± 8.4 mm month^{-1}) while, at inter-annual scale, the highest total annual evaporation value was obtained in 2019 (1880 mm yr^{-1}) and followed by years 2020 (1805 mm yr^{-1}), 2009 (1781 mm yr^{-1}) and 2008 (1741 mm yr^{-1}) (Table 4.8).

The monthly mean (±SD) rainfall in Lake Baringo basin varied from 16 ± 63.8 mm month^{-1} in January to 132 ± 149.0 mm month^{-1} in August for the period from 2008 to 2020 (Table 4.8). At annual scale, the total annual precipitation varied from 225 mm yr^{-1} in 2009 to 3707 mm yr^{-1} in 2020 indicating that 2009 was the dry year while, 2020 was the wet year for the period 2008 to 2021 (Table 4.8b).

4.2.3 Inflow and loss Components

Table 4.9 summarizes the mean (± SD) annual values of all water inflow and loss components of the lake water balance model for two periods 1970-1995, and 2008-2021. The results show that the inflows into the lake for the period 1970-1995 consisted of 927 ± 223 mm yr^{-1} of direct rainfall and 2,675 ± 202 mm yr^{-1} from the surface runoff, together giving a total inflow of 3,602 mm yr^{-1} of storage water in the lake. This storage was lost through evaporation on an average of 2,138 ± 238 mm yr^{-1}, abstractions of water from inflows for agricultural purposes (1,050 ± 123 mm yr^{-1}), and underground seepage losses (246 ± 48 mm yr^{-1}), thus totaling to outflow component of 3602 mm yr^{-1} with left storage of 168 mm yr^{-1}. However, the mean (± SD) annual values of the principal water balance components for the period 2008-2021 indicated that the mean rainfall over the lake was 1209 ± 143 mm yr^{-1}, mean evaporation was 1719 ± 58 mm yr^{-1}, discharge in terms of underground seepage and abstracted water for agricultural purposes were 328 ± 24 mm yr^{-1} and 128 ± 19 mm yr^{-1}, respectively, while surface runoff was 2904 ± 159 mm yr^{-1}. At the end of monitoring in June 2021, the storage of water in the lake was estimated to be 1938 mm yr^{-1} arrived by runoff (Table 4.9).

The rainwater directly falling to the lake contributed 25% of the total inflow while, the runoff was estimated at 75% to the total inflow for the period 1970-1995 but the storage was less and estimated at about 5% (Table 4.9). Furthermore, the findings show that losses due to evaporation are estimated at 5% of the total losses while, seepage and irrigation losses are evaluated at 7% and 29% of the losses, respectively. During the period 2008-June 2021, 71% of the inflow was received from surface runoff, and only 29% was added by the direct rainfall water (Table 4.9) although the absolute storage volume contributed by the precipitation was about ten times more (1,938 mm yr^{-1}) than that of 168 mm yr^{-1} during the period 1970-1995. However, evaporation losses and supplemental irrigation accounted for 42 and 8% of the total outflow from the lake storage, respectively, and only 3% was lost through underground seepage with a storage estimation of 47% (Table 4.9). The residence time of the water in the lake has two-fold decreased from 3.1 years during the period 1970-1995 to 1.8 years for the period 2020-2021 indicating an increase in water movement in the lake from a slow movement to a rapid increase in lake water renewal time, with the increase of inflow rates in the lake over time (Table 4.9).

Table 4.9: Inflow and outflow components for the water balance model of Lake Baringo for two periods 1970-1995 and 2008-2021. Water balance components of the lake are in mm, except the residence time which is in years.

Water balance components	**Quantity**			
	Period 1970-1995	Proportion (%)	Period 2008-2021	Proportion (%)
Rainfall (mm)	927±223	25.0	1209±143	29.4
Surface runoff (mm)	2,675±202	75.0	2,904±159	70.6
Total inflow	**3,602±345**	**100**	**4,113±148**	**100**
Evaporation losses (mm)	2,138±238	59.3	1,719±58	41.8
Abstractions (mm)	1,050±123	29.2	328±24	8.0
Seepage losses (mm)	246±48	6.8	128±19	3.1
Lake storage (mm)	168	4.7	1,938	47.1
Residence time (R_t) (year)	**3.1**	-	**1.8**	-
Total outflow	**3,602**	**100**	**4,113**	**100**

4.2.4 Sensitivity of Lake levels to meteorological variables

Figure 4.8 shows the sensitivity of the lake levels to variability in air temperature, cloud fraction, rainfall, and relative humidity. The horizontal dashed lines correspond to the current lake level (9.5 m) and the vertical dashed lines represent the present conditions (current values) of the corresponding meteorological variable.

The results showed that changes in the meteorological variables have high and relative effects on the lake water levels (Figure 4.8). For exemple, temperature increase from present value will lead to decreased lake levels (Figure 4.8a) with a 1.6°C temperature increase from current conditions (25.4°C) resulting in lake level less than 5 meters while a 2 °C decrease leads to possible lake Overflow. Temperature increase from its current value will lead to high lake evaporation (Figure 4.8a). A temperature increase of around 1.6°C from present conditions (25.4°C) leads to the lake's evaporation increase from 1715 mm yr^{-1} to 1745 mm yr^{-1}, about 30 mm yr^{-1} evaporation increase/decrease for every 6% increase in temperature.

Sensitivity analysis shows changes in rainfall to result into changes in lake level (Figure 4.8b). For instance, a decline of more than 50% (around 650 mm yr^{-1}) in annual average precipitation from current value of 2,707 mm yr^{-1} would lead to dry-out conditions leading to lake level less than 2 m, while a rise of about 9 % (245 mm yr^{-1}) in the annual mean rainfall would lead to lake overflow (1.6 m increase in lake level) (Figure 4.8b). Inversely, the effects of rainfall on the lake evaporation in the lake area seem to be less pronounced and may mainly occur beyond 3500 mm yr^{-1} (Figure 4.8b). Lake levels are also relatively sensitive to changes in relative humidity and cloud cover fraction (Figures 4.8c and 4.8d) in Lake Baringo. However, the increase in relative humidity does not affect lake levels significantly (Figures 4.8c) while, evaporation declines with increased relative humidity.

Only around 2.5% increase in cloudiness from present conditions results in Lake Overflow with around 1.3 m lake level rise with slight effect on lake evaporation (Figures 4.8d).

The reduction in both relative humidity and cloud cover fraction by up to 64% for relative humidity and 20% for cloud cover, respectively, does not lead to the lake level decrease below 4 m. For example, the effect of changing relative humidity on lake evaporation occurs from 60% to 80% and would cause lake evaporation to slightly drop by about 20 mm yr^{-1} (Figure 4.8d).

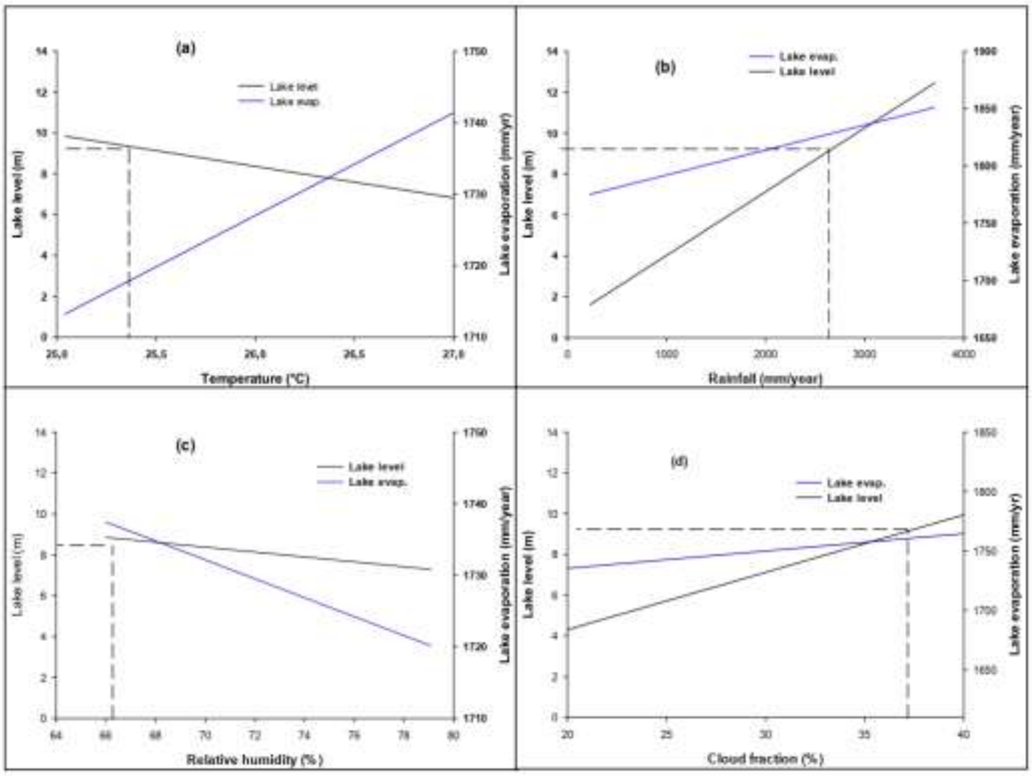

Figure 4.8: Lake water level sensitivity to changes in (a) temperature; (b) rainfall; (c) Relative humidity; (d) cloudiness for the period January 2008-June 2021.

4.3 Objective 3: Effects of lake water level fluctuations on water quality variables and lake fisheries

4.3.1 Temporal variation in lake properties

The PCA ordination results showed a clear separation between the years based on the lake water quality properties and fisheries yields (landings) for a 13-year time frame (2008-2020) (Figure 4.9). Of the five axes extracted in the PCA, only axes 1 and 2 are presented as they explain the majority of the extracted variance at 81.25%, and 12.59%, respectively.

The years 2013 and 2014 ordinated in the upper left of the biplot were characterized as having high TN levels and total fish yields while, the years from 2008 to 2012 are ordinated to the lower right and characterized by higher turbidity levels fluctuating between 73.69-185 NTU, higher WQI values ranging between 860.19 and 1443.47, higher TP concentrations varying from 146.66 to 240.35 µg L^{-1} indicating periods of poor water quality and high yearly water level variations measured as annual amplitudes (1.56-9.36 m). Besides, the years from 2015 to 2020 that were ordinated in the upper and lower left of the biplot, respectively, are marked by higher DLTM, indicating an increase in the annual lake water levels relative to the long-term average. The other water quality variables such as DO and temperature, known to affect assemblages and lake function, were not strong descriptors in the grouping of years based on the first two components that majorly explained the grouping results of the years at 93.84% (Figure 4.9).

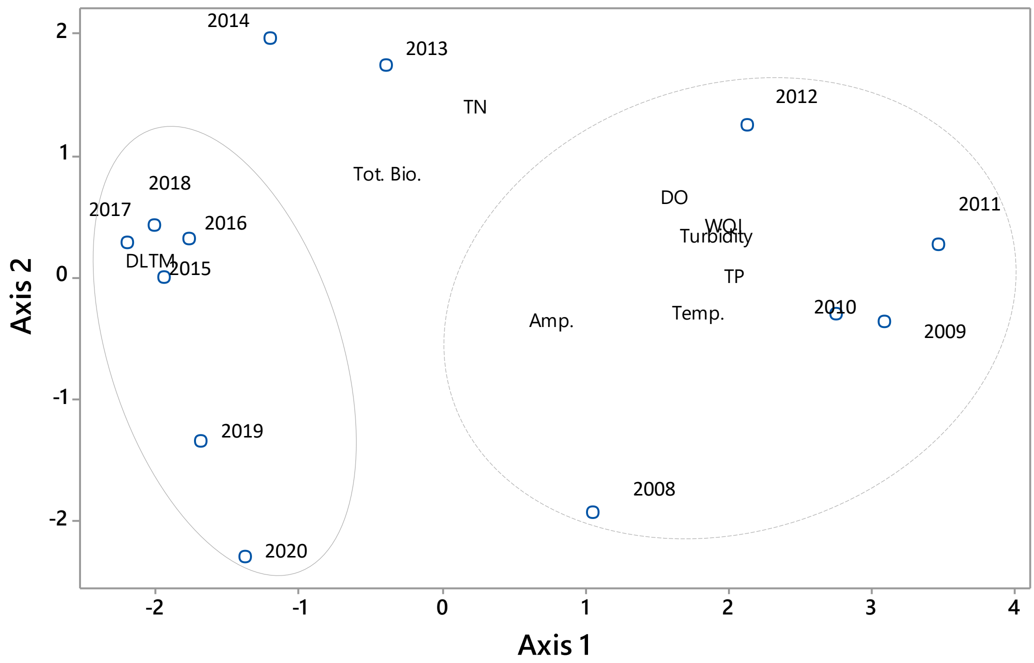

Figure 4.9: PCA analyses of Lake Baringo used to characterize water-level fluctuation influences on its fishery production and water quality properties from 2008-2020. Amp.: amplitude, Temp.: temperature, Tot. Bio.: total biomass and DLTM: is the annual difference from the long-term mean or average lake level.

4.3.2 Water Level Fluctuation Patterns

The lake water levels fluctuated by yearly amplitude (WLamp ± SD) of 2.25 ± 2.00 m from 1956 to 2020 (Figure 4.10). Consequently, a LOWESS plot showed that the lake water level amplitude fluctuated during the period 1956 to 1975 and then declined steadily up to its lowest level in 1989 (amplitude = 0.1 m), with a subsequent increase to peak amplitude in 2008 (9.4, m) before a decreasing fluctuation between 2008 and 2021 (Figure 4.10). Linear regression analysis showed a non-significant (p = 0.12) negative relationship between yearly amplitude and time while, the waveform Sine 3 parameter modeled as; Wlamp =2.328*sin (2*π*year/43.45+6.28), unlikely revealed a significant result ($r^2 = -0.64$) even though $p < 0.05$ but $r^2 = -0.64$ indicating a poor fit than a horizontal line.

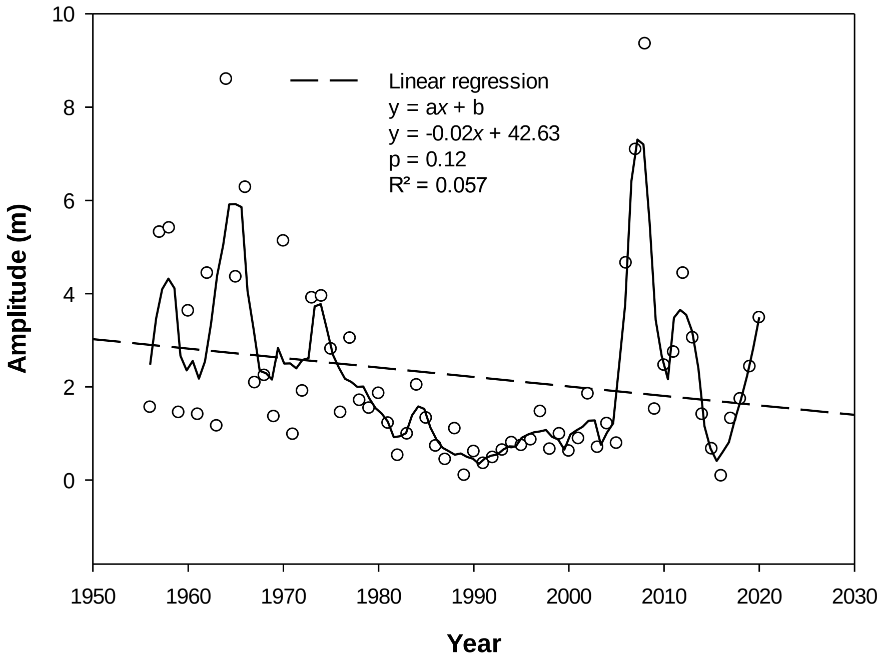

Figure 4.10: Long-term (1956-2020) patterns in yearly lake level amplitudes for Lake Baringo, Kenya. The continuous line represents a LOWESS fit to the data.

The trends in water level variations as measured by DLTM (Figure 4.11) were poorly explained by linear regression model (p = 0.1229, r^2 = 0.059). However, the waveform Sine (3 parameter) model: DLTM = 1.376*sin (2*π*year/(42.98 + 6.28)), was highly significant although with a weak fit ($p < 0.0001$, $r^2 = 0.21$), and indicated peak rise in Lake Baringo water levels after every ~20 years (Figure 4.11).

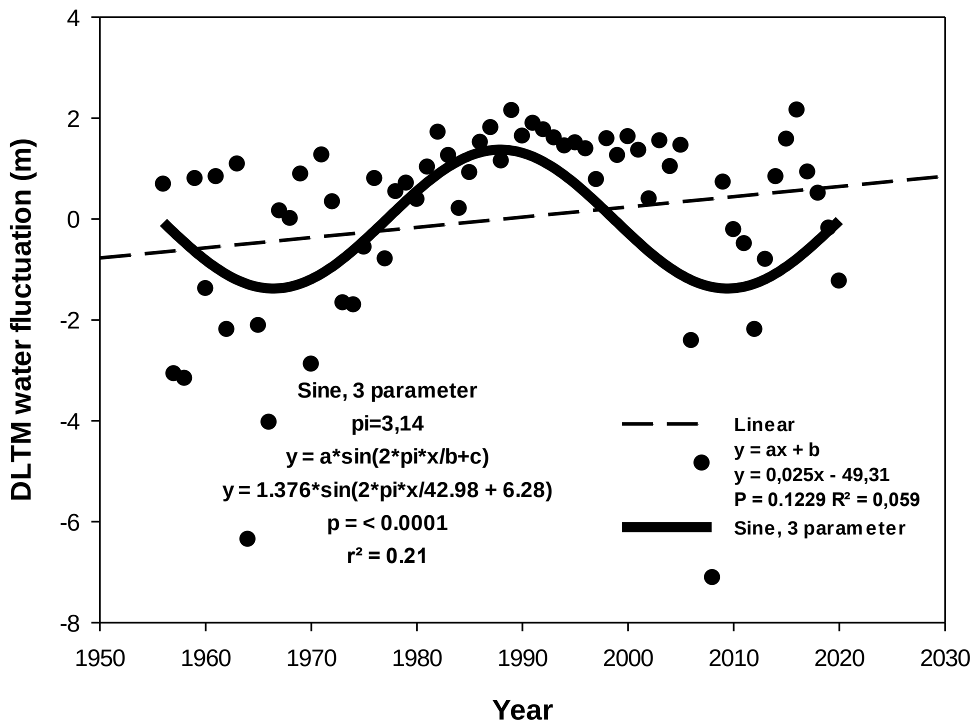

Figure 4.11: Linear and waveform regressions of mean Water Level Fluctuations, expressed as the difference from the long-term mean (DLTM) over time (1956-2020) for Lake Baringo, Kenya. The positive trend suggests that water levels are rising over time; a 20-year periodicity in oscillation is discerned.

The depth frequency plot (Figure 4.12a) showed that for 25% of the years (1956-2020), the average depth of the lake was about 2 m while, for 20% of the years, the average depth was 3 m. For less than 10% of the years, the mean depth of the lake reached 4 m. Additionally, only for less than 5% of the years, the lake means depth ranged between 10-14 m during the period from 1956-2020. The lake depth-area relationship (Figure 4.12b) showed significant dependence (R^2 = 0.74) of the lake depth on its area (A, km^2) modeled as: depth = $0.179e^{0.019A}$; Figure 4.12b. Thus, in current conditions, a 1 m increase in lake depth leads to about 90 km^2 increase in lake area with likely influence on the riparian communities through overflows.

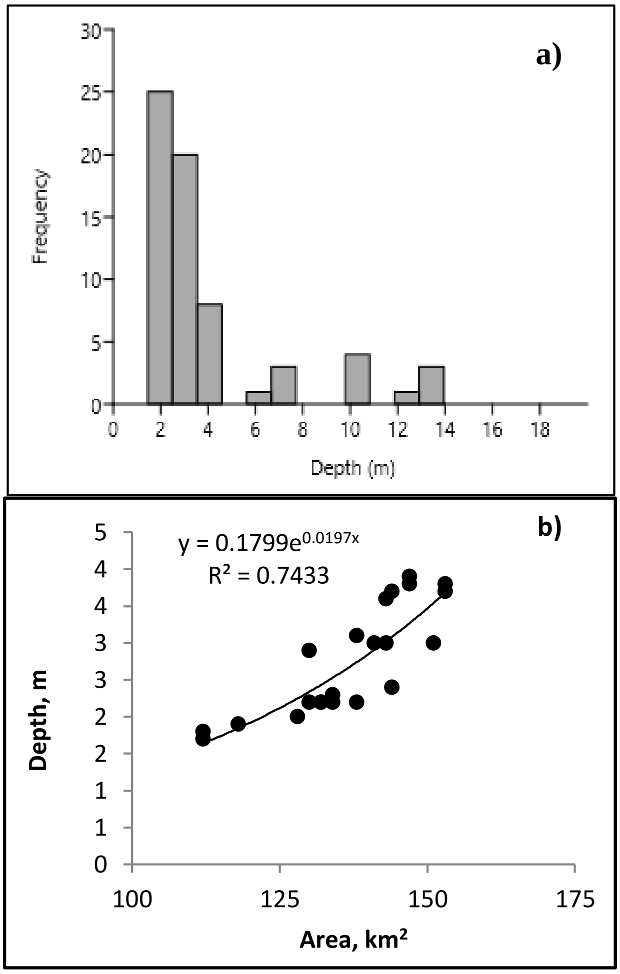

Figure 4.12: a) Depth-frequency distributions displayed as a histogram of pixel depth and b) Lake water level-surface area relationship of Lake Baringo, Kenya, measured from 1956-2020.

4.3.3 Relationship between lake level fluctuations (WLFs) and water quality parameters

A significant correlation ($p < 0.01$) was found between nine water quality parameters (conductivity, depth, TP, PO_4^{3-}, WQI, DO, temperature, turbidity, and NO_3^-) and DLTM (Table 4.10) for the long-term (2008-2020) data. Only, one variable (SiO_4^{4-}) was significantly ($p < 0.01$) correlated with the yearly amplitude (WLamp) in Lake

Baringo indicating the DLTM to better predict the changes in water quality in the lake at an inter-annual scale. Except for the depth and different forms of nitrogen (NO_3^-, NO_2^- and NH_4^+) that had significant negative correlations with DLTM, other water quality variables demonstrated positive concordance with increasing DLTM over time in the lake (Table 4.10). NH_4^+ that varied between 21.75 to 48.8 µg L^{-1}, also showed significant negative correlation ($p < 0.05$, r = -0.64) with DLTM over time in the lake. Silicate ions (SiO_4^{4-}) ranged from 18.92 to 27.43 mg L^{-1} across all years (2008-2020). SiO_4^{4-} did not demonstrate a concordance with DLTM but showed a positive significant correlation with yearly amplitude of lake levels ($p = 0.008$, r = 0.70) indicating the ion is tracked by short-term changes in the lake. There was no significant concordance between Chl-a, and TN with any of the WLFs indices over time in the lake. The water quality index (WQI) as a measure of the lake water quality was significantly correlated with the DLTM ($p = 0.001$, r = 0.80) and not the lake water level amplitude, WLamp (Table 4.10).

At an intra-annual scale, a significant correlation ($p < 0.01$) was found between six water quality parameters (conductivity, depth, Secchi disc, temperature, NO_2^- and NO_3^-) (Table 4.10) and monthly amplitude (WLamp) (Table 4.11) for the monthly data collected from January 2020 to June 2021.

Only, one parameter (NH_4^+) was significantly ($p < 0.01$) correlated with the DLTM in Lake Baringo indicating the WLamp to be a better predictor of water quality in the lake at an intra-annul scale. Except for the Secchi disc which had a strong significant positive correlation with DLTM and a moderate significant negative correlation with monthly amplitude (Table 4.11), other parameters demonstrated very weak concordance with both monthly DLTM and WLamp over the short time period (2020-

2021) in the lake. TN ranging from 2.83 µg L^{-1} in December 2020 to 71.32 µg L^{-1} in April 2020, DO varied between 4.44 mg L^{-1} in May 2020 and 9.48 mg L^{-1} in December 2020 and pH ranged from 7.09 December 2020 to 9.92 in January 2020 all showed high potential concordance ($p < 0.05$) but weak ($r < 0.50$) with the monthly lake level amplitude (WLamp) over time.

Table 4.10: The Pearson product-moment correlation coefficient (r) between selected variables describing the water quality parameters and hydrological indices describing the water level fluctuations (WLFs) in Lake Baringo (Kenya) between 2008 and 2020.

Hydr. variables	Pearson	WQI	Cond. (µs/cm)	DO (mg/L)	Tur.(NTU)	Depth (m)	Temp. (°C)	Chl.a (µg/L)	TP (µg/L)	PO_4^{3-} (µg/L)	NO_2^- (µg/L)	NO_3^- (µg/L)	TN (µg/L)	SiO_4^{4-} (mg/L)	NH_4^+ (µg/L)
DLTM	r	0.80**	0.84***	0.74**	0.74**	-0.88***	0.72**	-0.25	0.92***	0.84***	-0.67*	-0.75**	0.10	0.08	-0.64*
	P	0.00**	0.00***	0.00**	0.00**	7.71E-05***	0.01**	0.41	6.6E-06***	0.00***	0.01*	0.00**	0.75	0.79	0.02*
WLamp	r	0.46	0.36	-0.13	0.40	-0.37	0.36	0.20	0.26	0.06	-0.32	-0.20	-0.30	0.70**	0.09
	P	0.11	0.22	0.67	0.18	0.21	0.22	0.52	0.39	0.84	0.28	0.51	0.33	0.01**	0.78

Values with *** highlight strong significant concordance ($p < 0.001$), ** highlight significant concordance ($p < 0.01$) while * highlights potential concordance ($p < 0.05$). DLTM: Difference from the long-term mean, WLamp: water level amplitude.

Table 4.11: The Pearson product-moment correlation coefficient (r) between selected variables describing the water quality parameters and hydrological indices (DLTM and WLamp) describing the water level fluctuations (WLFs) in Lake Baringo, Kenya, between January 2020 and June 2021.

	DLTM		WLamp	
Variables	***p***	**r**	***p***	**r**
Chl.a (µg L^{-1})	0.53	-0.21	0.40	0.16
TN (µg L^{-1})	0.44	-0.51	0.03*	0.20*
TP (µg L^{-1})	0.41	-0.21	0.40	0.21
SiO_4^{4-} (mg L^{-1})	0.95	0.13	0.61	0.02
NO_3^- (µg L^{-1})	0.17	-0.66	0.00**	-0.34**
NO_2^- (µg L^{-1})	0.78	-0.79	0.00***	-0.07***
PO_4^{3-} (µg L^{-1})	0.13	0.27	0.28	-0.37
NH_4^+ (µg L^{-1})	0.01**	0.04**	0.89	0.59
WQI	0.06	0.25	0.33	0.45
Depth (m)	0.59	0.61	0.01**	0.14**
Temp. (°C)	0.76	0.68	0.00**	0.08**
EC (µs/cm)	0.07	-0.60	0.01**	-0.45**
DO (mg L^{-1})	0.86	0.55	0.02*	-0.04*
TDS (mg L^{-1})	0.76	0.21	0.40	-0.08
SD (cm)	0.04*	0.81*	4.2E-05***	-0.49***
pH	0.29	-0.48	0.04*	-0.27*

Values with *** highlight strong significant concordance (p < 0.001), ** highlight significant concordance (p < 0.01) and * highlights potential concordance (p < 0.05). DLTM: Difference from the long-term mean, WLamp: water level amplitude.

The Gaussian 4 parameter unimodal model further provided a significant fit ($p < 0.05$) to the relationship between WQI and DLTM ($R^2 = 0.89$) and not with the lake amplitude (Figure 4.13a and b) in the lake. These results indicated that a range of DLTM of ≥ 2 m and in the range of 6.5 to ≥ 10 m provided a good water quality of the lake as WQI approached the WHO recommended value of WQI ≤ 100, while DTLM values of ≤ 2 m resulted into poor water quality. The Gaussian, 4-parameter model: WQI = 314.68 + 1252.05*exp(-0.5*((x - 3.88)/2.2)^2), yielded a significant relationship (p = 0.0024) between WQI and DLTM (R^2 = 0.89) (Figure 4.13a). These results indicated that the water quality index (WQI) was lowest in years closest to the highest long-term mean (DLTM = 9.9 m) and that WQI increased with decreasing lake water levels (Figure 4.13b).

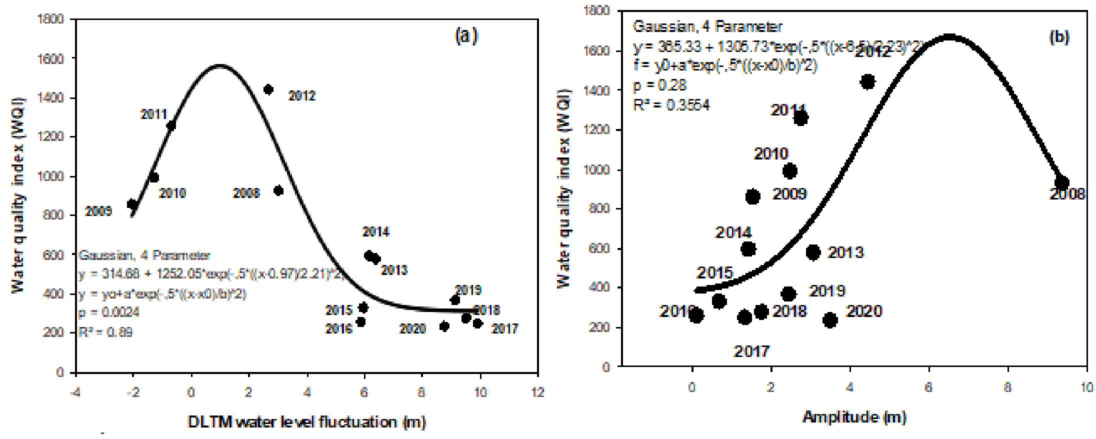

Figure 4.13: Gaussian model fit demonstrating unimodal response of water quality index (WQI) with Water Level Fluctuations indices over a 13 year period (2008-2020) for Lake Baringo, Kenya.

At the intra-annual scale, the Gaussian 4 parameter unimodal and linear models performed on the monthly data collected from January 2020 to June 2021 did not provide a significant fit (p > 0.05) to the relationship between WQI and DLTM and with the lake amplitude (Figure 4.14 a and b). The results indicate that at a monthly scale, there is no significant relationship between DLTM and water quality. However, the very dry months of January and February provided a good water quality of the lake as WQI approached the WHO recommended value of WQI ≤ 100.

The Gaussian, 4-parameter model: WQI = 543.897 + 432.48*exp(-0.5*((DLTM-0.18)/0.17)^2) and linear model; WQI = 398.039 + 264.94*A, yielded weak relationships (p < 0.1) between WQI and DLTM (p = 0.0966, R^2 = 0.354) (Figure 4.14a) and between WQI and Amplitude, A (p = 0.062, R^2 = 0.202) (Figure 4.14b) using monthly data for the period January 2020-June 2021. These results also indicated that in general the water quality index (WQI) was lowest in months closest to the highest long-term monthly mean (DLTM = 1.3 m) and increased with decreasing lake water levels. Also, the findings indicated that the water quality index (WQI) was lowest in months closest to the lowest water level amplitude and increased with increasing monthly lake level fluctuations for the period 2020-2021.

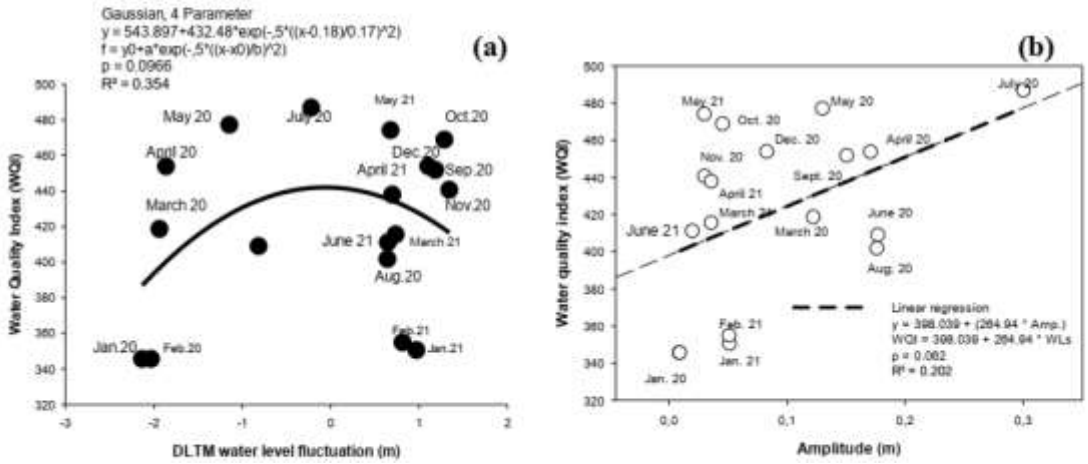

Figure 4.14: (a) Gaussian and (b) linear models fit demonstrating both unimodal and linear responses of water quality index (WQI) to Water Level Fluctuations indices (DLTM and Amplitude) over 18 months (January 2020- June 2021) for Lake Baringo, Kenya.

4.3.4 Relationship between lake level fluctuations and fisheries

The annual fisheries yields from 1982-2020 (years of available data) ranged from approximately 8 metric tons in 1994 to 496 metric tons in 2017 averaging close to 227 metric tons per year (Figure 4.15). Linear regression analysis highlighted a significant relationship ($p < 0.05$, $R^2 = 0.66$) between annual fisheries yields (landings) and the water level variations (WLs) indexed as annual amplitude indicating the direct effect of WLFs on the lake's fisheries production modeled as Fishery yield = 78.15 + (29.94 * WLs); $R^2 = 0.66$ (Figure 4.15). These results suggest that water level changes have an influence on fish catchability in the lake at inter-annual time scale.

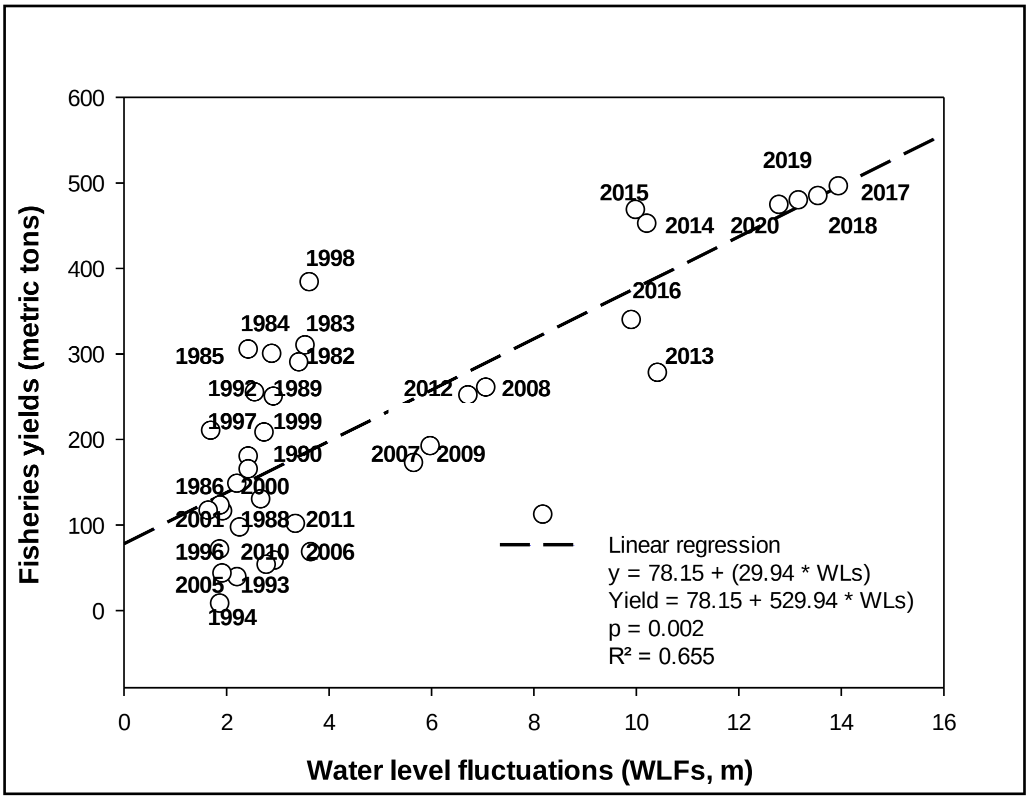

Figure 4.15: Relationship between annual mean fisheries landings and water level fluctuations in Lake Baringo, Kenya, from 1982-2020.

The fishery variables (fish yields and average annual fish condition factor) were correlated with the WLFs indicators using Pearson's correlation coefficients (r) shown in Table 4.12. The results demonstrated a strong significant positive relationship ($p < 0.001$) between the condition factor of the endemic tilapia species, *Oreochromis niloticus baringoensis*, with the yearly lake water level amplitude (r = 0.69). However, there were potential significant correlations (r = ~ 0.5) between the condition factor of the lungfish, *Protopterus aethiopicus* with DLTM (r = 0.48), and between the annual yield of the barb, *Labeobarbus intermedius*, and DLTM (r = 0.50) (Table 4.12). These results indicated positive and negative relationships for *P. aethiopicus* and *L. intermedius* with mean WLFs, respectively. There were no significant relationships between either DLTM or amplitude (WLFs) and total biomass as well as with fish species yields in the lake except for *L. intermedius* (Table 4.12).

Table 4.12: The Pearson product-moment correlation coefficient (r) between hydrological variables describing the water level fluctuations (WLFs) and fisheries ofLake Baringo, Kenya, between 2008 and 2020.

Variables	Pearson correlation	Relative condition factor (kr)				Fisheries yields				
		O. n	***P. a***	***C. g***	***B. i***	**Tot. Bio.**	***P. a***	***O. n.***	***C. g***	***L. i***
DLTM	r	0.045	0.478*	0.454	0.173	0.127	0.264	-0.030	-0.106	-0.499*
	P	0.884	0.098*	0.119	0.573	0.680	0.383	0.922	0.730	0.082*
WLamp	r	0.691***	0.084	-0.148	0.228	0.215	0.244	0.096	0.442	-0.358
	P	0.008***	0.785	0.630	0.453	0.481	0.421	0.756	0.130	0.230

Values with *** highlight significant concordance ($p < 0.001$), while values with * highlight potential concordance ($p < 0.10$); DLTM: Difference from the long-term mean; Tot. Bio.: total biomass; *P. a.:Protopterus aethiopicus; O.n.: Oreochromis niloticus baringoensis; C.g.: Clarias gariepinus; and L.i.:Labeobarbus intermedius.*

The monthly fisheries yields ranged from approximately 8.40 metric tons in September 2020 to 12.35 metric tons in June 2021 averaging close to 9.94 metric tons per month (Figure 4.16). Gaussian, 4-parameter model fit showed a significant relationship ($p < 0.05$) between monthly fisheries yields and the water level fluctuations (WLFs) (monthly amplitude) indicating the direct effect of WLFs on the lake's fisheries production (Fishery yield = 9.38 + 6.46*exp(-0.5*((WLs - 13.4)/0.26)²); R^2 = 0.44; Figure 4.16). These results suggest that water level changes have an influence on fish catchability in the lake on both monthly and annual scales.

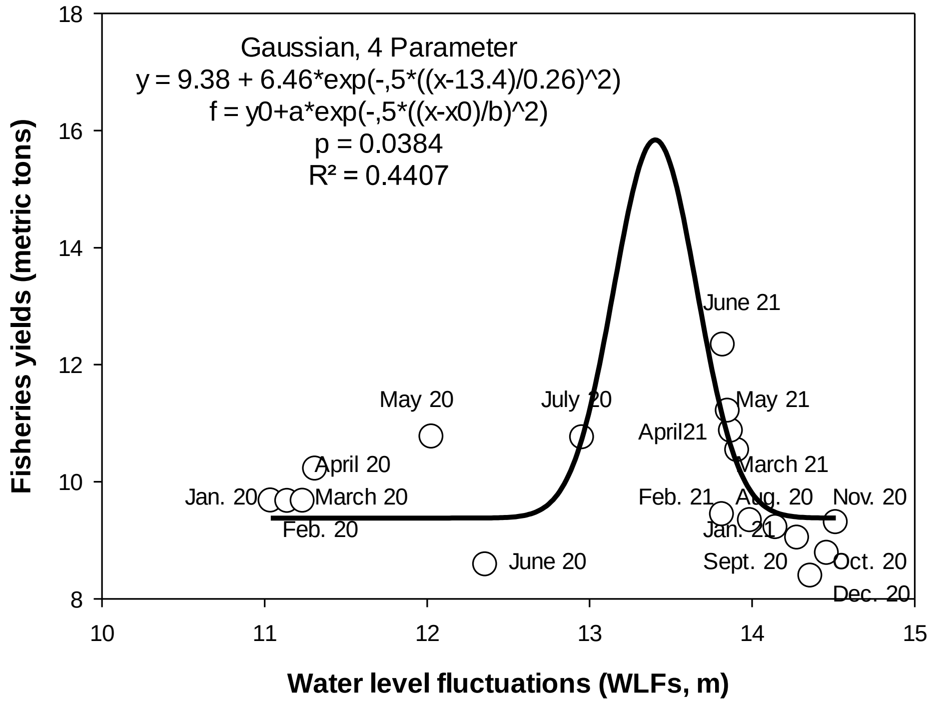

Figure 4.16: Gaussian model fit demonstrating unimodal response of fisheries yields with Water Level Fluctuations over a 18 month period (January 2020-June 2021) for Lake Baringo.

4.4 Objective 4: Modeling the food web properties and fisheries dynamics in Lake Baringo using Ecopath mass-balanced model

4.4.1 Catch composition and Trends in commercial fish landings

Lake Baringo's commercial fish landings by weight varied considerably from 1982 to 2020 (Figure 4.17)**.** The individual landings comprised four main species namely: *Oreochromis niloticus baringoensis, Labeobarbus intermedius, Clarias gariepinus* and *Protopterus aethiopicus* which are currently the key commercial fish landings in Lake Baringo**.** The total fish landings (total catch and individual landings) exhibited variable trends among years with the lowest value of 8 tons yr^{-1} in 1994 to a peak of 455 tons yr^{-1} in 2000 (Figure 4.17)**.**

The *O. n. baringoensis* landings from the lake have fluctuated from 199 tons/yr in 1982 to 28 tons yr^{-1} in 2000 showing a decreasing trend from 1980 to 2020 after reaching a peak of 326 tons yr^{-1} in 1990. *Clarias gariepinus* total annual landings have increased from 28 tons yr^{-1} in 1982 to 120 tons yr^{-1} in 2000, and then decreased to 10 tons yr^{-1} in 2010 before fluctuating with decreasing trend from 2011 to 2020. The total annual landings related to *Labeobarbus intermedius* ranged between 46 tons yr^{-1} in 1982 to a peak of 198 tons yr^{-1} in 2014 before decreasing to 10.1 tons yr^{-1} in 2020. Annual catches of African lungfish, *Protopterus aethiopicus*, exhibited a general increase in landings by weight from its lower values of 11 tons yr^{-1} in 1984 (year when *P. aethiopicus* was for the first time reported in the commercial catch in Lake Baringo). The highest landings for *P. aethiopicus* were 199 tons yr^{-1} and 251 tons yr^{-1} obtained in 1999 and in 2014, respectively, while the lowest landing was 1 ton yr^{-1} observed in 1994 (Figure 4.17).

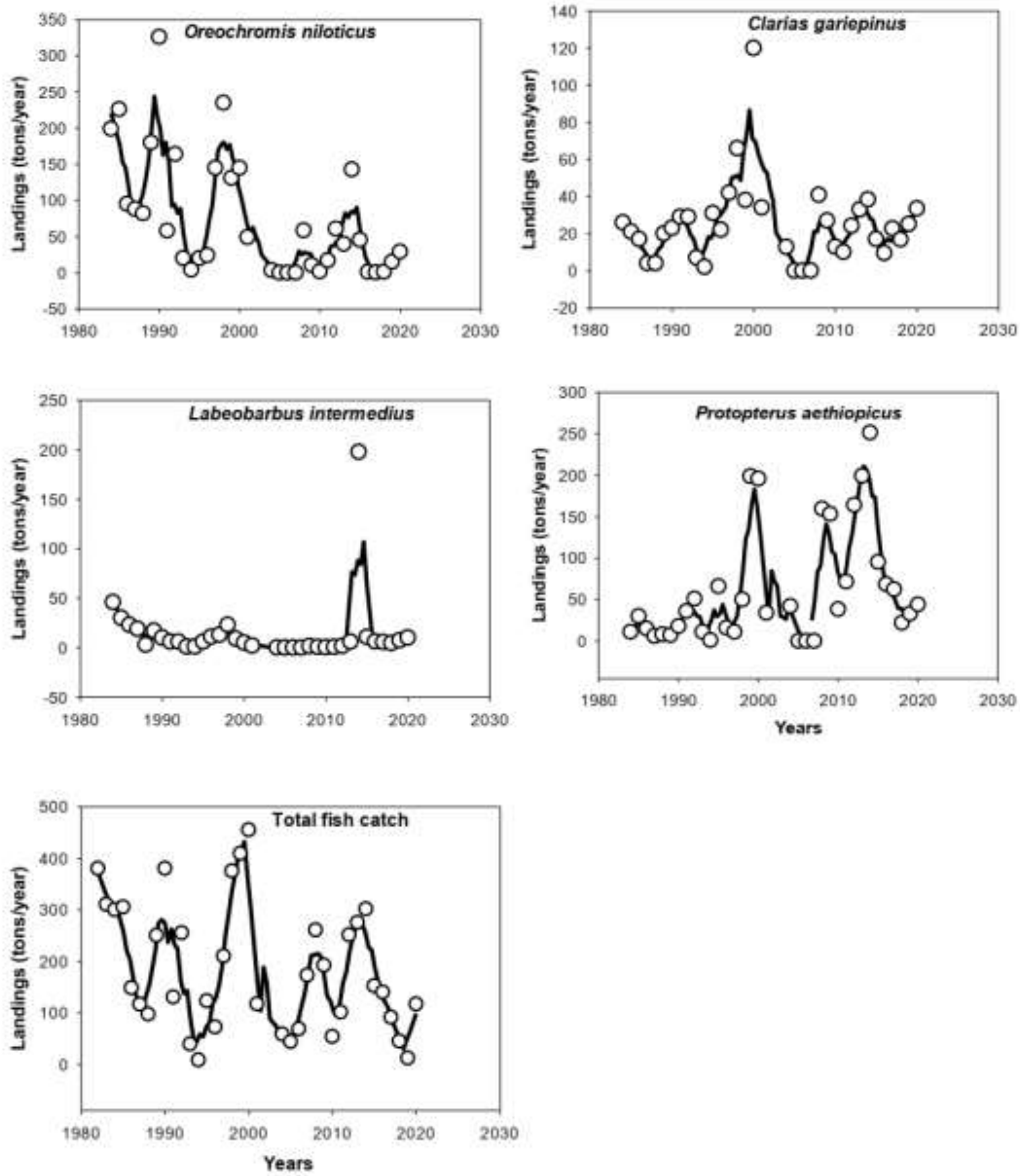

Figure 4.17: Time-series of commercial fish landings in Lake Baringo, Kenya. Circles are mean values of all samples collected monthly from 1982 to 2020. Solid line is LOWESS trendline of monthly mean values.

4.4.2 EcopathTrophic flow models for Lake Baringo

4.4.2.1. Model structure

The trophic models of the lake generated for 1999, 2010 and 2020 are shown in Figure 4.18, while the data used to generate the trophic models are shown in Table S3 (Appendix 2). The models indicate that the lake contains three main trophic levels in each of the three time periods. The size of bubbles in the three models (Figure 4.18) represents the biomass proportions of each functional group in the lake implying, bottom-up controls from high biomass of the phytoplankton in the models. The vertical axis in the three models (1999, 2010, 2020) represents an approximate trophic position for each functional group, with producers (phytoplankton) and detritus low on this axis, and predators near the top. The trophic positions (TL) are shown on Table 4.13. The horizontal axis (Figure 4.18) represents different sources of producers located on the left side of the axis with detritus on the right side of the axis in the three trophic models. Phytoplankton represented a large and stable source of stored production which supplied energy to many species compared to detritus whose biomass was lowest in the 1999 and 2020 trophic models (Figure 4.18a,c). There are clear trophic groupings in the three trophic models (1999, 2010, 2020) based on the role of each functional group, with phytoplankton, detritus, zooplankton, mollusks and exploited fish species separated into different trophic levels (TLs).

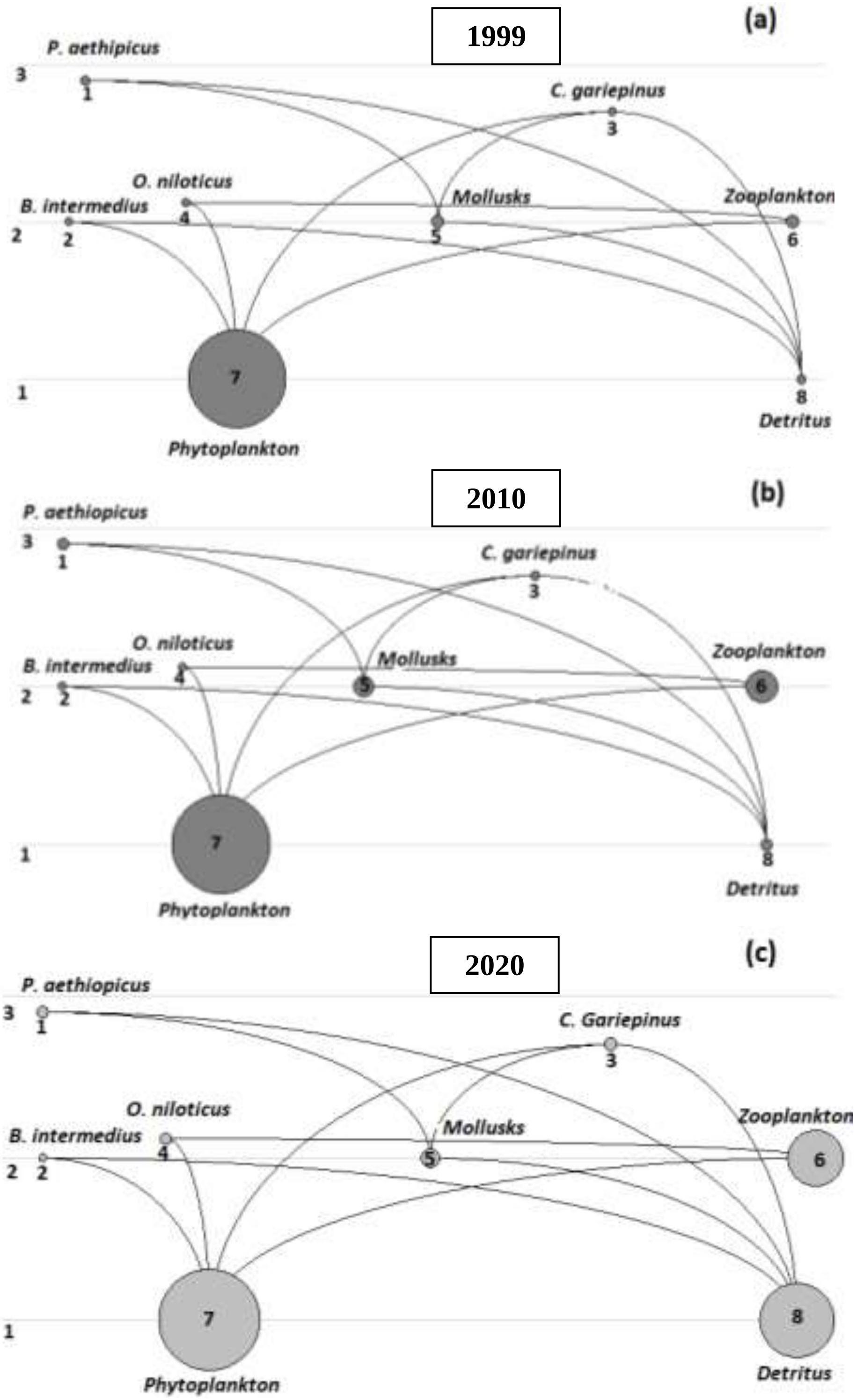

Figure 4.18: Trophic models for Lake Baringo's fishery during (a) 1999, (b) 2010 and (c) 2020. Vertical position approximates the trophic level. The circle size is proportional to the compartment (population and/or functional groups) biomass (g wet weight m^{-2}).

The highest trophic level (TL) in the three trophic models (1999, 2010, 2020) of the lake was occupied by *Protopterus aethiopicus* (TL= 2.9, Table 4.13, Figure 4.18) while, the lake ecosystem was dominated at the base (trophic level TL1) by the flows involving phytoplankton and detritus. *Labeobarbus intermedius*, *Oreochromis niloticus baringoensis*, mollusks and zooplankton as primary consumers were concentrated at trophic level TL2 while, *Clarias gariepinus* was in-between trophic levels TL2 and TL3 for the three years (1999, 2010, 2020) (Figure 4.18). The highest trophic level was 2.9 for *P. aethiopicus* followed by 2.7 for *C. gariepinus*, values that are between 2.5 and 3.0 belonging to high-order primary consumers or primary carnivores (Table 4.13).

The low-order primary consumers were *O. n. baringoensis*, *B. intermedius*, mollusks, and zooplankton with trophic levels ranging between 2 and 2.12, while the class of primary producers was constituted by phytoplankton and detritus at trophic level TL1. These three trophic model visualizations describe (Figure 4.18) the overall ecological network of Lake Baringo in time.

Biomass estimates for phytoplankton and *Labeobarbus intermedius* (an omnivore) were the largest and smallest, respectively, for the three trophic models (1999, 2010, 2020) in the lake implying bottom-up controls (Figure 4.18, Table 4.13).

Table 4.13: Final parameters for ecopath models representing the Lake Baringo's fishery for 1999, 2010 and 2020, where TL = trophic level, B = biomass reported in wet weight, P/B = turnover rate, Q/B = consumption rate, EE = ecotrophic efficiency, and P/Q = gross efficiency. Bold figures (TL, EE and P/Q) are outputs from the Ecopath model.

Groups/species name	TL	B (tons/km^2)	P/B (year)	Q/B (year)	EE	P/Q (year)
Periods (years) 1999						
1 *P. aethiopicus (PA)*	**2.90**	0.376	0.325	42.00	**0.000**	**0.008**
2 *L. intermedius (LI)*	**2.00**	0.003	0.024	0.34	**0.000**	**0.071**
3 *C. gariepinus (CG)*	**2.70**	0.211	0.128	23.00	**0.000**	**0.006**
4 *O. niloticus (ON)*	**2.12**	0.073	0.325	8.00	**0.000**	**0.044**
5 Mollusks (Mo)	**2.00**	1.230	15.856	13.70	**0.903**	**0.15**
6 Zooplankton (Zoo)	**2.00**	1.120	0.769	12.54	**0.081**	**0.061**
7 Phytoplankton (Phy)	**1.00**	72.880	0.863		**0.247**	
8 Detritus (Det)	**1.00**	0.350			**0.312**	
Periods (years) 2010						
1 *P. aethiopicus (PA)*	**2.90**	0.324	0.420	42.12	**0.000**	**0.010**
2 *L. intermedius (LI)*	**2.00**	0.001	0.012	0.13	**0.000**	**0.095**
3 *C. gariepinus (CG)*	**2.70**	0.098	0.527	12.74	**0.000**	**0.041**
4 *O. niloticus (ON)*	**2.12**	0.025	0.767	3.25	**0.000**	**0.236**
5 Mollusks (Mo)	**2.00**	1.230	13.330	159.90	**0.802**	**0.084**
6 Zooplankton (Zoo)	**2.00**	2.600	2.966	338.00	**0.001**	**0.088**
7 Phytoplankton (Phy)	**1.00**	27.375	35.100		**0.915**	
8 Detritus (Det)	**1.00**	0.320			**0.637**	
Periods (years) 2020						
1 *P. aethiopicus (PA)*	**2.90**	4.500	0.714	98.00	**0.000**	**0.137**
2 *L. intermedius (LI)*	**2.00**	0.62	0.013	13.00	**0.000**	**0.199**
3 *C. gariepinus (CG)*	**2.70**	4.650	1.010	101.00	**0.000**	**0.125**
4 *O. niloticus (ON)*	**2.12**	3.590	0.780	78.00	**0.000**	**0.080**
5 Mollusks (Mo)	**2.00**	12.230	64.980	106.83	**0.913**	**0.132**
6 Zooplankton (Zoo)	**2.00**	8.480	184.860	1848.60	**0.002**	**0.120**
7 Phytoplankton (Phy)	**1.00**	327.83	7146.600		**0.669**	
8 Detritus (Det)	**1.00**	152.70			**0.001**	

4.4.2.2. Ecotrophic Efficiency (EE) Patterns

The output results (Table 4.13) indicate that the estimated *Ecotrophic Efficiency (EE)* values are < 1 for the three models and did not vary considerably for the different functional groups in the lake, ranging from 0.00 to 0.903 in 1999, 0.00 to 0.915 in 2010, and from 0.00 to 0.913 in 2020 (Table 4. 13). The *EE* values for the top predators (*P. aethiopicus* and *C. gariepinus*) and other fish species were zero indicating that the top predators are not consumed within the system. The *EE* values for the primary producers (phytoplankton and detritus) were slightly high, compared to the secondary producer (zooplankton) in the three models (Table 4.13) suggesting that the primary producers are more consumed in the system than the secondary producers – a bottom-up regulation. The EE values of mollusks in the three trophic models (1999, 2010, 2020) were high approaching 1 suggesting that mollusks were highly consumed or insufficient in the system's food web of the lake.

4.4.2.3. Biomass distributions of functional groups

The biomass distribution of the different functional groups depicted in Figure 4.18 is summarized in Table 4.13. The detritus biomass and phytoplankton biomass (measured indirectly as chlorophyll-a concentration) exhibited alternating trends compared to the fish total biomass over the three time periods. From 1999 to 2010, the biomasses showed 9% decline from 0.35 to 0.32 g wet wt m^{-2} for detritus and 62.4 % decline from 72.8 to 27.4 g wet wt m^{-2} for phytoplankton, before peaking in 2020 (152.7 g wet wt m^{-2} for detritus and 327.8 g wet wt m^{-2} for phytoplankton) (Table 4.13). In 2020 trophic model, the detrital biomass was the highest indicating their influence on nutrient and energy cycling in the lake. The zooplankton biomass presents increasing trends comparable to those of the total biomass over the three time periods (Table 4.13). The highest biomass of zooplankton (84.8 g wet wt m^{-2}) and the lowest biomass (1.12 g wet wt m^{-2}) were observed in 2020 and 1999, respectively (Table 4. 13). The zooplankton biomass patterns were quite similar to those of phytoplankton from trophic model in 1999 to trophic model in 2020 except in 2010 trophic model where phytoplankton biomass decreased. Generally, the year 2020 (characterized by high water level in the lake due to heavy rains) exhibited higher biomasses for both phytoplankton (327.8 g wet wt m^{-2}), zooplankton (8.4 g wet wt m^{-2}), as well as for the fish biomass (0.45 g wet wt m^{-2}) from the lake than years 1999 and 2010 with light rain where the phytoplankton (27.38 g wet wt m^{-2} in trophic model 2010), zooplankton (1.12 g wet wt m^{-2} in trophic model 1999) and fish biomass (0.324 g wet wt m^{-2} in trophic model 2010) were very low (Table 4. 13).

The estimated total fish biomass of the lake was 0.66 metric ton km^{-2} yr^{-1} in 1999, 0.45 metric ton km^{-2} yr^{-1} in in 2010, and 1.34 metric ton km^{-2} yr^{-1} in 2020, with *P. aethiopicus* and *C. gariepinus* exhibiting higher biomasses than the other fish groups except in 2020 when *C. gariepinus* had higher biomass followed by *P. aethiopicus*. The principal species with a great increment of biomass were *O. n. baringoensis* (from 0.073 g wet wt m^{-2} in 1999 to 0.359 g wet wt m^{-2} in 2020), and *C. gariepinus* (from 0.098 g wet wt m^{-2} in 2010 to 0.465 g wet wt m^{-2} in 2020). It is important to note that the biomasses of all functional groups from the producer to consumers decreased in the year 2010 except for zooplankton which increased from 1.120 g wet wt m^{-2} in 2010 to 84.80 g wet wt m^{-2} in 2020.

Table 4.14 summarizes the distribution of the total biomass in different trophic levels in Lake Baringo's fishery system as per the models (Figure 4.18). The total biomass in different trophic levels in Lake Baringo varied from 31.97 g m^{-2} in 2010 to 568.62 g m^{-2} in 2020. The biomasses of functional groups at the base of the trophic level for the three trophic models (73. 23 g wet wt m^{-2} in 1999, 27.69 g wet wt m^{-2} in 2010 and 480.5 g wet wt m^{-2} in 2020; Table 4.14) were high (as expected), compared to the other levels in the three trophic models.

The proportion of biomass at trophic level TL1 decreased over time from 96% in the 1999 trophic model to 84.5% in 2020 whereas it increased at trophic levels TL2 from 3.3% in 1999 model to 15.23% in 2020 model and TL3 from 0.7% in 1999 trophic model to 1.1% in 2010. However, at trophic level TL3 the biomass decreased from 1.1% in 2010 to 0.30% in 2020. In contrast, from 1999 to 2020, the proportion of biomass in the upper trophic levels (II) increased marginally. However, all types of biomass flows into the system showed a decline from 2010 to 2020. Most of these flows (> 80%) were concentrated in trophic level I for the three models resulting in a decreasing average transfer efficiency from 6.4% in 1999 to 0.5% in 2020 after reaching a value of 4.7% in 2010 (Table 4.14). The trophic level III characterized by lowest biomasses peacked in 2020 (1.51 g m^{-2}) after decreasing from 0.50 g m^{-2} in 1999 to 0.36 g m^{-2} in 2010, while the trophic level IV was absent in the three trophic models of the lake indicating the lack of secondary carnivores in the system.

Table 4.14. Proportional total system biomass (g wet weight m-2) for discrete trophic levels (TL) from Lake Baringo ecosystem models for years 1999, 2010, and 2020.

TL	Biomass (g/m^2)	Biomass (%)
Periods (years) 1999		
I	73.23	96.00
II	2.52	3.30
III	0.50	0.70
IV	0.00	0.00
Total	**76.25**	**100**
Periods (years) 2010		
I	27.69	86.60
II	3.92	12.30
III	0.36	1.10
IV	0.00	0.00
Total	**31.97**	**100**
Periods (years) 2020		
I	480.50	84.50
II	86.61	15.23
III	1.51	0.30
IV	0.00	0.00
Total	**568.62**	**100**

4.4.3 Network analysis and Ecosystem indices

4.4.3.1 Ecological functioning indicators

The output parameters of the three annual trophic models (Figure 4.19) of Lake Baringo are summarized in Table 4.15. The network indices used to assess the ecosystem status included TPP/TR, TPP/TB, TB/TST, CI, and Finn's cycling index (see section 3.5.5.4 in methods). The relatively high productivity with regards to the total biomass and respiration (TPP/TB and TPP/TR) ratios (Table 4.15) indicates the lake is geologically young and ecologically immature, respectively (sensu Odum, 1969).

Table 4.15 shows that there has been an increasing trend in productivity in relation to biomass and respiration, an increasing trend in food web connectivity (measured by food web Connectivity Index; CI) from 0.22 in 1999 to 0.30 in 2010 and then to a lower value of 0.23 in 2020. The System Omnivory Index (SOI, an index of food web diversity) showed a decreasing trend from 0.07 in 1999 to 0.05 in 2020. There was a decreasing trend in ecological transfer efficiency (TE) and nutrient cycling rate (measured by Finn's cycling index) (with a reduction in number of groups that energy flow passes through as shown by reduction in Finn's cycling index) from 1999 to 2020 (Table 4.15). Concerning the food web organization descriptors; both system's Ascendency (A) (an indicator of ecological activity) and Overhead **(**O) (measure of energy in reserve of an ecosystem to resist perturbations or resilience) decreased twofold and fourfold, respectively, between 1999 and 2010, and increased considerably between 2010 and 2020 indicating the increasing potential of resistance of the ecosystem to disturbances (Table 4.15).

The total system throughput (TST) that indicates flow network complexity and describes network lengths in a food web was 84% lower in 2010 than in 1999, attaining its greatest value in 2020 indicating that the network lengths and flow networks in the lake's food web were stronger in 2020 than in 1999 and 2010 (Table 4.15). Additionally, all other parameters showed similar trends as TST from 1999 to 2020 except B/TST (an indicator of energy flows in the food web) that had an opposite trend, and ∑Q and ∑R which showed increased trends among the models (Table 4.15).

Ascendency (an indicator of the potential for development or ecological activity) demonstrated an increasing trend in the potential for the lake to recover from perturbation over the last two decades (Table 4.15). The Finn's cycling index (FCI) (defined as the percentage of the total flow in the food web) decreased from 1999 to 2020 indicating the decrease in nutrient cycling rates; whereas the system overhead (Ov) (energy in reserve of an ecosystem indicating the potential for recovery and ecosystem stability) and system capacity (C) decreased from 1999 to 2010 and increased from 2010 to 2020 indicating the fluctuating nature of the potential of the lake to recover from impacts or perturbations.

The system omnivory index (SOI) and the connectance index (CI) are also used as indicators of ecosystem maturity and are expected to be higher in late stage of maturity in ecosystems (mature systems) (Odum, 1971). The SOI observed in Lake Baringo for the three trophic models was 0.07 in 1999, 0.06 in 2010 and 0.05 in 2020 (Table 4.15), a reducing trend, suggesting reduced prey organisms in the diets and reduced prey diversity in the system. The connectance index (CI) defined as the ratio of the actual links between groups to the numbers of the possible theoretical links, was estimated at 0.22 in 1999, 0.30 in 2010 and 0.23 in 2020 (Table 4.15) indicating annual variation in the food web complexity of Lake Baringo.

Net system production (NSP) (defined as the difference between the total primary production and total respiration), another ecosystem maturity indicator, is higher in immature ecosystems and close to zero in mature ones. The three Ecopath models (1999, 2010, 2020) in Lake Baringo calculated very high NSP values (t. km^{-2}. yr^{-1}) of 5950.99 in 1999; 964.46 in 2010 and 239.537 in 2020 (Table 4.15) highlighting the decreasing trend in NSP suggesting the progressive shift of the ecological status of the lake from ecological immaturity status towards an ecological maturity stage (equilibrium between prey-predator biomasses) of Lake Baringo's ecosystem.

The gross efficiency (GE) of the fisheries in the lake was 0.013 in 1999, 0.033 in 2010 and 0.0018 in 2020, all the gross efficiency values were < 1 indicating that the three trophic models in Lake Baringo to be mass-balanced over time (Table 4.15).

Table 4.15: Summary output statistics of the mass-balance Ecopath model of Lake Baringo showing ecosystem attributes; network flow indices, and information flow Indices during 1999, 2010, and 2020. The trend indicates the direction towards a mature system (± indicating increasing/decrease respectively). N/a stands for parameters associated with fisheries.

Parameters (unit)/periods		Trend	1999	2010	2020	% Change 1999-2010	% Change 2010
Sum of all consumption (g m^{-2} yr^{-1})	∑Q	+	181	913	13,035.34	4.04	1426.58
Sum of all exports (g m^{-2} yr^{-1})	∑EXP	N/a	59,509.9	9,644.6	239,537.20	-0.84	247.36
Sum of all respiratory flows (g m^{-2} yr^{-1})	∑R	+	187	360	5,252.35	0.93	1458.11
Sum of all flows into detritus (g m^{-2} yr^{-1})	∑FD	+	5,951.91	965.75	239,539.80	-0.84	247.03
Total system throughput (g m^{-2} yr^{-1})	TST	+	11,902.83	1,935.74	486,859.92	-0.84	250.51
Sum of all production (g m^{-2} $year^{-1}$)	∑P	+	5,952.43	971.77	249,965.50	-0.84	256.23
Total net primary production (g m^{-2} yr^{-1})	TPP	+	5,949.12	960.86	234,284.80	-0.84	242.83
Net system production (g m^{-2} yr^{-1})	NSP	+	5,950.99	964.46	239,537.20	-0.84	247.36
Total net primary production/total respiration	TPP/TR	-	3182.33	266.93	44.61	-0.92	-83.82
Total net primary production/total biomass	TPP/TB	+	7,839	3,036	56,428	-0.61	1758.63
Total biomass/total throughput	TB/TST	+	0.0001	0.0002	0.0000	1.57	-94.52
Total biomass (exc. detritus) (g m^{-2} yr^{-1})	TB	+	75.90	31.65	415.19	-0.58	12.12
Gross efficiency (Actual catch/primary production)	GE	-	0.0128	0.0329	0.0018	1.58	-94.52
Network flow indices							
Throughput cycled (exc. detritus) (g m^{-2} yr^{-1})	TCI	N/a	0.00	0.00	0.00		
Throughput cycled (inc. detritus) (g m^{-2} yr^{-1})	TCId	+	124.90	101.90	419.30	-0.19	3.11
Finn's cycling index (% of total)	FCI	-	0.0105	0.0044	0.00086	-0.58	-0.80

Total number of pathways (no.)	#P	N/a	6.00	6.00	6.00	0.00	0.00
Food web connectance (dimensionless)	CI	-	0.22	0.30	0.23	0.36	-0.23
Omnivory index (dimensionless)	OI	-	0.07	0.06	0.05	-0.14	-0.17
Mean transfer efficiency (%)	TE	-	6.43	4.72	0.49	-0.27	-0.90
Information flow indices							
Ascendency (total) flowbits	A	+	4,706.81	1,941.37	49,638.24	-0.59	24.57
Overhead (total) flowbits	Ov	+	9,028.60	2,122.90	65,163.96	-0.76	29.70
Capacity (total) flowbits	C	+	15,678.21	8,082.8	157,001.2	-0.48	18.43
Ov/C (%)	Ov/C	+	57.58	26.26	41.51	-0.54	58.07
A/C (%)	A/C	+	30.02	24.02	31.62	-0.20	31.64
Pedigree index	P	-	0.269	0.115	0.115	-0.57	0.00

4.4.3.2 Fisheries and Keystoneness indicators

The fisheries indicators used included primary production required (PPR, the unit of primary production over total ecosystem production), mean trophic level of landings (TLc), and the gross fishing efficiency (GFE), and showed significant differences between the three Ecopath models (1999, 2010, 2020) of the lake (Table 4.15; Figure 4.19 a,b,c). The results show, for instance, that the mean trophic level of the catch (TLc, which quantifies the mean trophic position of exploited organisms) increased from 1999 to 2010 and then decreased from 2010 to 2020 (an overall decline) indicating that the organisms were more exploited in 1999 and 2010 but less exploited in 2020 (Figure 4.19a), a general fishing-down of the food web. The trophic level of the catch was lowest in the 2020 model and highest in the 2010 model (Figure. 4.19a), indicating that the fishing intensity changed with time in Lake Baringo suggesting high fishing impacts between years 1999 and 2010 and low fishing impacts during the year 2020 in the lake.

The results also showed that the primary production required (PPR, an indicator of the sustainability of fisheries in terms of energy) was low and fluctuating over time in the lake (Figure 4.19b). The highest value was 3.5% in 2020, while, the lowest value was 0.025% in 2010 suggesting a decreased sustainability in 2010 but low PPR indicating overall sustainability of the fisheries.

The gross fishing efficiency (GFE, which measures the degree to which the fish stocks are exploited in the ecosystem) was high in 2010 and very low in 2020 suggesting that the fish stocks are underexploited in 2020 compared to other years (1999 and 2010) (Figure 4.19c).

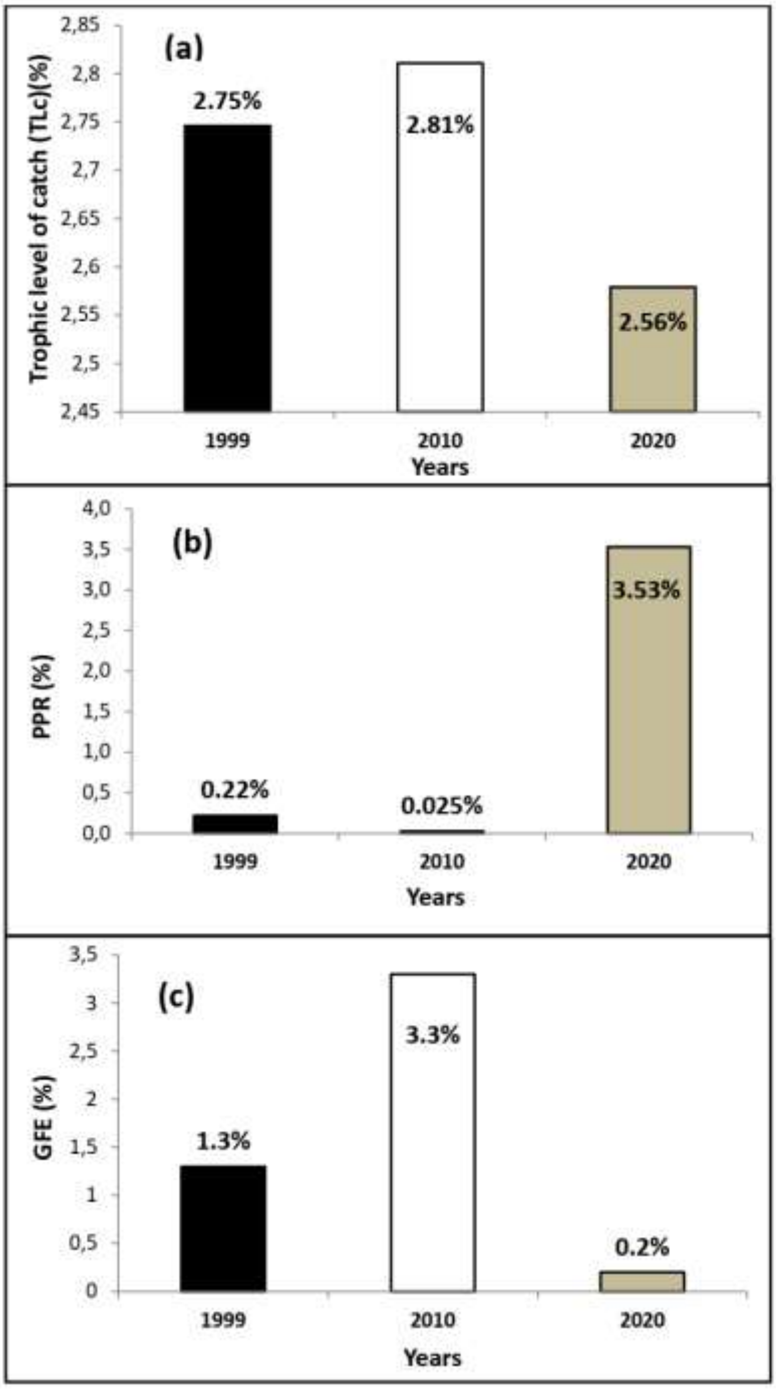

Figure 4.19. Fisheries indicators (trophic level of the catches; TLc, primary production required; PPR and gross fishing efficiency; GFE) calculated for three different trophic models for the food web of Lake Baringo, Kenya.

Figure 4.20 shows the keystoneness index and the relative trophic impact of the different functional groups in the 1999, 2010 and 2020 Ecopath models of Lake Baringo. For each functional group, the keystoneness index (*y*-axis) is reported against the overall effect (*x*-axis). Overall effects are relative to the maximum effect measured in each trophic web; thus, for the *x*-axis, the scale is always between 0 and 1. The functional groups are represented by number corresponding to species keystoneness rankings in the model.

The keystone index (KSi) (an indicator of system controls by groups) values from the three trophic models (1999, 2010, 2020) indicated the functional group with the most keystoneness properties to be *O. n. baringoensis* followed by zooplankton group for the three generated trophic models of the lake (Figure 4.20). This result indicated that *O. n. baringoensis* is the species that controls the population of species in Lake Baringo playing both roles of an omnivorous feeder and prey of carnivores in the lake's food wed. The next ranks in keystoneness in Lake Baringo are occupied by mollusks, *Clarius gariepinus* and *Protopterus aethiopicus* in 1999 trophic model, while in 2010 and 2020 models, the next keystone species are mollusks and *Protopterus aethiopicus*, respectively (Figure 4.20). However, only *O. n. baringoensis* had low biomass and highest relative trophic impact in the three trophic models (1999, 2010, 2020) of Lake Baringo, indicating that the *O. n. baringoensis* is the main keystone species in the lake.

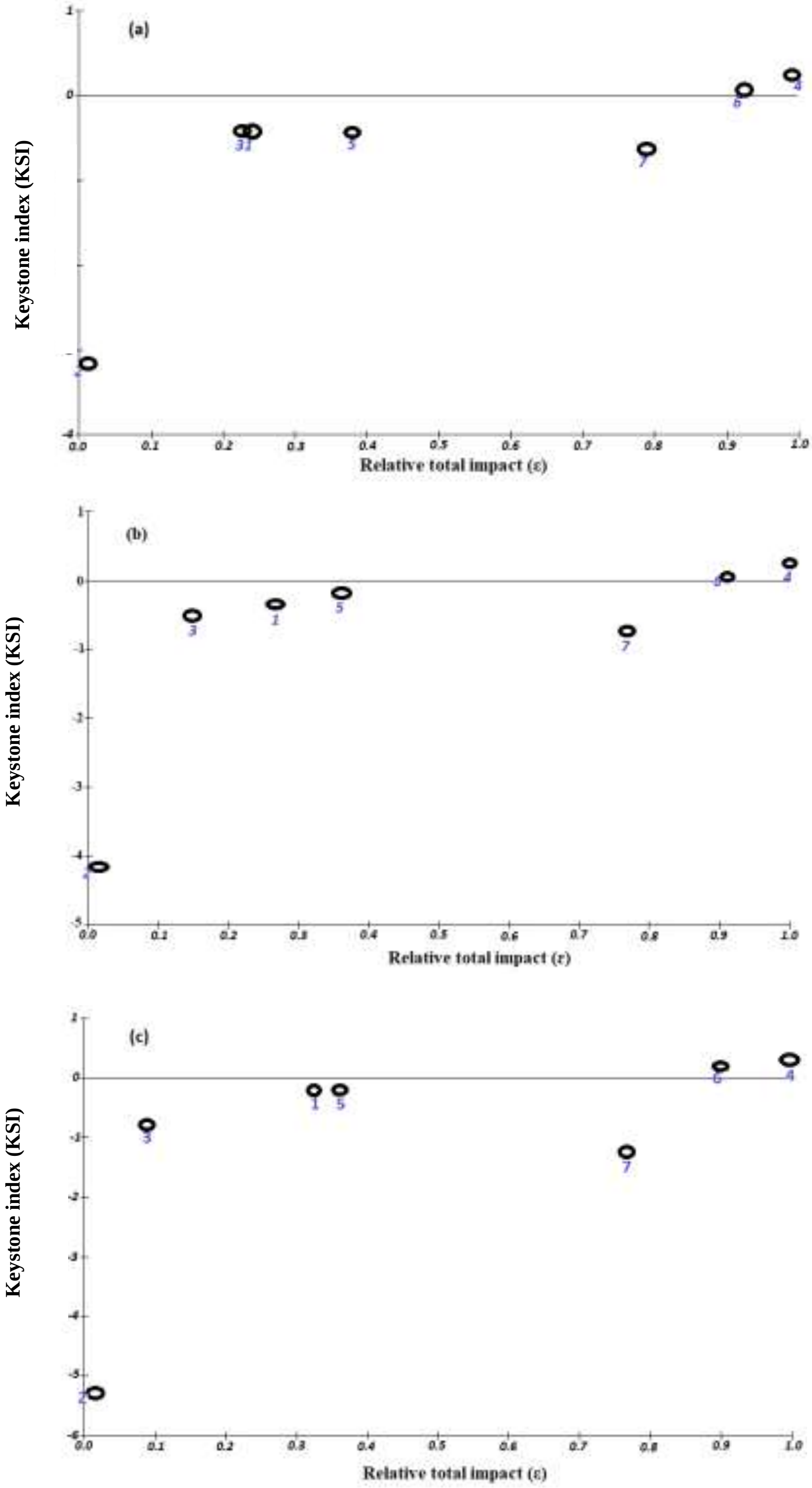

Figure 4.20: Relative overall effect (εi) and keystoneness (KSi) of each species/functional group from the Lake Baringo ecosystem models for (a): 1999, (b): 2010, and (c): 2020. The number corresponds to keystoneness rankings from 1999 species/functional groups: 1-*P. aethiopicus*; 2-*B. intermedius*; 3-*C. gariepinus*; 4-*O. n. baringoensis*; 5-Mollusks; 6-zooplankton and 7-phytoplankton.

4.4.4 Comparative analysis with African lakes

The outputs obtained from the Ecopath models in Lake Baringo water was compared with those from some other Afrotropical lakes (Table 4.16). The total consumption (∑Q) for the lakes: Lakes Volta (Ghana), Awassa (Ethiopia), Malawi (Malawi), Kariba (Zimbabwe) and Victoria (Tanzania) were quite similar, but lower compared to Lake Baringo trophic model in 2020 and higher compared to Lake Baringo trophic models in 1999 and 2010 (Table 4.16). The total production (∑P) for Lakes Baringo, Kariba and Awassa, however, was much higher, compared to Lakes Volta, Malawi and Victoria. This indicates a high productivity in Lakes Baringo, Kariba and Awassa that is not being fully utilized by the fish species. The total primary production (TPP) value (t km^{-2} yr^{-1}) of Lake Baringo waters is the highest (96,086 in 2010 and 23,428.48 in 2020) among the lakes; the second highest is Lake Kariba ecosystem, (Table 4.16). TPP/TR value describes the maturity of the ecosystem. The TPP/TR value of Lake Baringo waters is highest among the lakes but still shows the lake is more immature ecologically compared to others. The connectance index (CI) and system omnivory index (SOI) values reflecting the complexity of the relationships among food weds due to the number of links between functional groups suggested that the food web complexity of different trophic levels in the lake was low compared to Lakes Volta and Victoria. Compared to other lakes, the SOI of Lake Baringo (0.07 in 1999, 0.06 in 2010 and 0.05 in 2020) was quite similar to that of Lake Volta (0.06) but very low when compared to other African Lakes (Awassa, Malawi and Kariba) suggesting a relatively low degree of omnivory in the two lakes (Lakes Baringo and Volta) (Table 4.16).

Table 4.16: Comparison of system statistics estimated by Ecopath model for Lake Baringo and other tropical lakes.

Parameter	Lake Baringo (1999)	Lake Baringo (2010)	Lake Baringo (2020)	Lake Volta	Lake Awassa	Lake Malawi	Lake Kariba	Lake Victoria
Sum of all consumption (t km^{-2} yr^{-1}) (∑Q)	181	913	13,035.3	1,554.8	2,382.1	2,6	2,4	2,2
Sum of all exports (t km^{-2} yr^{-1}) (∑EXP)	5,950.9	9,644.6	2 ,395.4	2,686.2	6,808.1	950	6.6	672
Sum of all respiratory flows (t km^{-2} yr^{-1}) (∑R)	187	360	5,252.4	933.8	1,442.7	1,6	1,6	1,2
Sum of all production (t km^{-2} yr^{-1}) (∑P)	5,952.4	97,2	24,996.6	3,930.1	8,881.0	3,1	13,9	2,4
Total net primary production (t km^{-2} yr^{-1}) (TPP)	5,949.1	96,1	23,428.5	3,620.0	–	2,6	13,6	1,9
Total primary production/total respiration (TPP/TR)	3182.3	266.9	44.6	3.9	9.8	1.6	8.6	1.6
Net system production (t km^{-2} yr^{-1}) (NSP)	5,950.9	96,5	23,953.7	2,686.3	6,974.1	950	11,9	669
Total primary production/total biomass (TPP/TB)	7,9	3,0	56,4	57.1	28.7	66	45.7	22.2
Connectance Index (CI)	0.2	0.3	0.2	0.4	–	–	–	0.3
System Omnivory Index (SOI)	0.07	0.06	0.05	0.06	–	0.15	0.13	0.19

Source: Lake Volta (Mensah *et al*., 2019), Lake Malawi (Moreau, 1995), Lake Kariba (Moreau, 1997), Lake Awassa (Fetahi, 2005; Fetahi and Mengistou, 2007) and Lake Victoria: (Darwalla *et al*., 2010)

4.4.5 Prebal and model balancing diagnostics

The ecotrophic efficiency (EE), gross efficiency (GE) and biomass ratios were simultaneously used as model balancing criteria for the three trophic models generated in this study. The criteria are used to indicate if the models are sufficiently constrained physiolocally so that they are balanced ecologically and thermodynamically. The ecopath models are only valid if they are balanced and meet the model assumptions.

The results of ecotrophic efficiency (EE), biomass ratios, vital rates across trophic levels (TLs) and gross efficiency (GE) as balancing indicators are shown on Tables 4.13, 4.14 and 4.15. The results showed that the ecotrophic efficiencies (EE) were < 1.0 and the P/Q ratio was low and between 0.1 and 0.3 indicating the three ecopath models for Lake Baringo were balanced (Table 4.14). Generally, the ecosystem biomass decreased with increasing trophic level as expected by theory indicating that assumptions of the model (e.g. equilibrium situation) were not violated (Figure 4.21; Table 4.14) and the declining patterns formed a pre-balancing (Prebal) for the models.

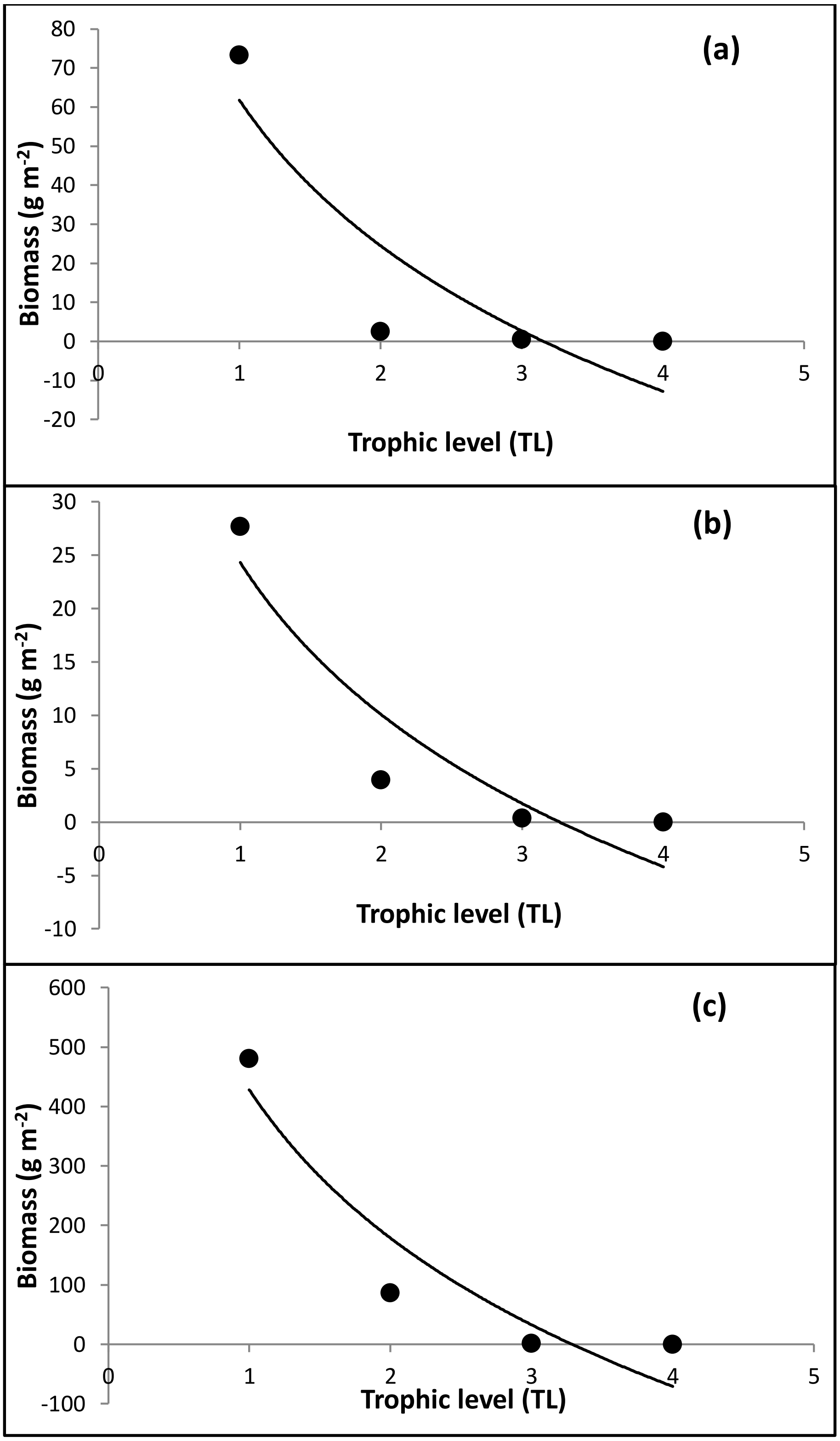

Figure 4.21: Relationship between system biomass values and trophic levels for functional groups in Lake Baringo, Kenya, for the years: (a): 1999, (b): 2010, and (c): 2020.

CHAPTER FIVE

DISCUSSION

5.1 Spatio-temporal variations in selected water quality parameters and lake trophic status

The results of this study provided for the first time, a long-term assessment of the water quality and trophic status of Lake Baringo, Kenya, using water quality indices. There were spatio-temporal differences in the physico-chemical properties of the lake. It is likely that the differential influences of shoreline inputs and location of the influent rivers resulted in differences in water properties at the various stations. Some water quality parameters (turbidity, fluoride, total phosphorus, SiO_4^{4-}, and DO) were found to be above the WHO (WHO, 2008; 2011) and APHA recommended thresholds (APHA, 2005) for livelihoods and ecological processes. For example, the dissolved oxygen (DO) values in all the stations were above the WHO recommended threshold of 5 mg. L^{-1} for human use of the lake water. Fluoride levels were above the WHO recommended threshold values of 1.50 mg. L^{-1} for human consumption indicating that the water in Lake Baringo is unsuitable as drinking water and requires pretreatment before its consumption by the riparian community. The high fluoride levels (2.0 – 13.0 mg L^{-1}) in Lake Baringo might have originated from natural (active volcanic eruption) and human (expansion of the irrigation area around the lake) processes, being one of the African Rift Valley Lakes which are reported to have high fluoride levels (Ayenew, 2008; Demelash *et al*., 2019). The effects of high fluoride concentration in natural water are the main cause of the development of dental fluorosis, responsible for brownish teeth (Edmunds and Smedley, 2013) and that is prevalent among the Lake Baringo riparian communities. Also, high fluoride levels expose the communities to cancer in extreme cases of exposition (> 7.5 mg L^{-1}) (Marshall, 1990).

Compared to international standard limits, DO, turbidity, fluoride, SiO_4^{4-}, and TN were above the interval limit values fixed for human consumption by the WHO (2008, 2011) and aquatic life processes by APHA (2005) and Rodier *et al.* (2009), reflecting the poor conditions of Lake Baringo waters over the years. Soluble silica (SiO_4^{4-}) and total nitrogen (TN) concentrations in the lake were above those threshold values recommended by APHA and WHO at 5 mg L^{-1} and 100 µg L^{-1}, respectively, while other nutrients (nitrates, phosphates, TP, and ammonium) loads in the lake showed variations that were within both the WHO and APHA recommended levels for ecological processes (APHA, 2005; WHO, 2008). Also, the turbidity values of the lake were found to be above the WHO permissible levels (5 NTU) for drinking (WHO, 2008) and for support of aquatic life for some species (Bartram and Balance, 1996; Rodier *et al.*, 2009; PNRM, 2009). High turbidity and total dissolved solids (TDS) have been reported to be the main physical indicators of water quality degradation in lakes (Wetzel, 2001) and may affect fisheries production and biodiversity (Aloo, 2002; Odada *et al*., 2006). In Lake Baringo, high turbidity has been reported to be a responsible factor that restricts the zooplankton abundance and diversity (Vörös and Padisák, 1991) which in turn affects the feeding habit of some fish species in the lake (Tarras-Wahlberg *et al.*, 2003; Omondi *et al*., 2014).

Strong positive correlations were found between lake turbidity and rainfall in the watershed, and between lake water turbidity and WQI indicating the influence of watershed erosion on the lake water quality. Baok (2007) reported that the correlation between turbidity and total suspended solids is very high in lakes and determines light reflectance and diffraction and hence the ecological processes. The negative relationship between Chl-a, and turbidity, found in this study, indicated the direct effect of turbidity on the light penetration into the water. High turbidity, therefore, reduces the productivity of the lake and is a likely cause of the low plankton diversity and reduced fisheries production in the lake (Tarras-Wahlberg *et al.*, 2003; Nyakeya *et al*., 2020). The high turbidity of the lake results mostly from the watershed erosion, the resuspension of the bottom sediments into the water column by the wind action, and partly from algal blooms (Oduor, 2003; Tarras-Wahlberg *et al.*, 2003; Omondi *et al.*, 2014). Recent attempts at watershed management (Nyakeya *et al*., 2020) have seen progressive temporal positive changes in some parameters as reported in this study. However, there is a need for more aggressive integrated watershed management interventions such as aforestation and land-use policy changes to reduce sediment input to the lake (Grobbelaar, 1984; Schagerl and Oduor, 2003). Holistic watershed management is required to sustain the water quality of the lake and fisheries productivity for livelihoods.

The WQI values found for Lake Baringo in this study were greatly above the WHO (2008) permissible limits (WQI=100) for drinking water indicating that the water of Lake Baringo is currently “unsuitable for human consumption” (WHO, 2008), despite the continued use by the local communities. However, the organic pollution index (OPI) values (4.5 – 4.9) were low and within the recommended limits for non-organically polluted lakes (Leclercq *et al*., 1987) indicating that the lake water is still organically unpolluted. Low organic nutrient loads are likely leached into the lake perhaps due to the less agricultural activities in the mostly semi-arid watershed. The ratio of N to P for the study period was less than 10 (OECD, 1982; Maberly *et al*., 200), suggesting that N is the limiting nutrient for the primary production in the lake (OECD, 1882). This result shows that the management of the nutrient enrichment in the lake has to prioritize the focus on P reduction rather than N because the N, fixation by certain cyanobacteria allows an escape from N limitation (Vollenweider, 1975; Schindler, 1978).

The trophic state of the lake as measured by the TSI (TP) and the Carlson scale (CTSI, Carlson, 1977) indicated that the lake's trophic status has been variable from being mesotrophic, hypereutrophic to eutrophic over the period 2008-2020 and also varies over a monthly scale. Consequently, Lake Baringo's trophic status has been fluctuating over time indicating the instability of the lake's ecological processes due probably to climate variability and human activities in the watershed. Similar variability in the trophic status of lentic systems has been reported elsewhere and attributed to anthropogenic influences (Al-Haidarey, 2016). These results indicate that the classification of the trophic status of Lake Baringo will depend on the methods used and the overriding factors within the watershed at a particular time. Implementation of land-use management policies at the watershed scale is needed for the lake water to support economic livelihoods of riparian communities and to sustain ecological processes.

5.2 Water balance modeling and its sensitivity to hydro meteorological variations

The hydro-climatic factors of the Lake Baringo Basin are similar to the regional climate characterized by a bimodal climate with rainy months (April to November) interrupted by dry months (December to March) driven by the twice-annual passage of the Intertropical Convergence Zone (Verschuren *et al*., 2009). The results on Lake

Baringo water balance components shows a strong similarity of the dynamics of precipitation in the basins of Rift Valley Lakes such as Naivasha and Turkana, indicating the same trends in both basins during heavy or slight rains. These observations concur with the results of Hulme *et al.* (2001) and Vandecasteele *et al*. (2010) who estimated that there was a wetting trend over the East-African region where some areas experienced increasing rainfall. The comparison of water levels between Lake Baringo and some other Rift Valley Lakes indicated that during 2002-2006, Rift Valley Lakes experienced low lake levels due to severe droughts related to the El-Niño Southern Oscillation cycle and forcing by the 2006 Indian Ocean dipole (Becker *et al*., 2010). Lake Baringo also experienced shrinking up with the water level estimated at 1.5 m leading to the closure of the fishing activities from 2002 to 2003 (Odada *et al*., 2006).

Currently (2020-2021), precipitation and river inflows represent 29.4% and 70.6%, respectively, of the overall inputs in the lake which is in disagreement with estimates in most Great African Lakes such as Lakes Kivu, Victoria, and Malawi where the precipitation contributes more than the runoff to the water balance (Bergonzini, 1998; Kumambala and Ervine, 2010; Muvundja *et al*., 2014). The evaporation losses and losses due to irrigation abstraction and underground seepage represented 41.8%, 8.0% and 3.1% of the total water losses, respectively, in Lake Baringo. These results show that the lake loses most of its water through evaporation. The annual precipitation variations demonstrated a significant (p = 0.034) positive relationship with yearly lake level with a lower correlation coefficient (R^2 = 0.081) due to probably some gaps in the lake water levels and precipitation data. These results concur with the recent findings of Taylor *et al*. (2013) that showed that the groundwater resources in East

Africa mostly depend on extreme rainfall rather than average rains. The gaps in lake levels is explained by the rising lake levels that make the water level gauges submerged during heavy rains while, the gaps in rainfall information is explained by both the scarcity of meteorological stations in the lower lake watershed and the poor maintenance of some meteorological stations in the lake basin.

The sensitivity analyses results showed that, in current conditions, the lake level is highly sensitive to changes in precipitation and evaporation, and moderately sensitive to air temperature, while less sensitive to relative humidity, and cloud cover fraction. This might be due to the estimate of some variables (eg. evaporation) which were not directly measured in the lake catchement. These results indicate that the water level will continue to drop following a decrease in the rainfall season and an increase in evaporation rates from the lake. These findings concur with those obtained by Kumambala and Ervine (2010) in Lake Malawi, Mbanguka *et al.* (2016) in Lake Babati, Muvundja *et al*. (2014) in Lake Kivu, and Borchardt and Trauth (2012) in lake Suguta where the lakes showed high sensitivity to changes in precipitation, except in Lake Tana where significant outflow led to the low sensitivity of the lake level to the precipitation (Kebede, *et al.*, 2006). Lake level sensitivity to temperature obtained in Lake Baringo has been witnessed in other East African lakes such as Lake Zaway in Ethiopia (Vallet-Coulomb *et al*., 2001) and Lake Victoria (Yin and Nicholson, 1998; Yin *et al*., 2000). For instance, temperature decline of about 2°C is projected to lead to lake overflow while the relative humidity has less impact on the lake level. Similar findings on sensitivity to relative humidity were reported in Lake Ziway where seemingly the relative humidity was found to have no significant impact on the lake level (Vallet-Coulomb *et al.*, 2001). The analyses showed the cloud cover to have less effect on the lake level demonstrating its lower influence on the evaporation rate in

Lake Baringo whereas its effect in Lake Nakuru-Elmenteita basin, Kenya, was found to be significant where 10% increase in cloudiness resulted in a 5.5% reduction of lake evaporation (Dühnforth, 2006). However, it is important to note that the calculation methods and assumptions may affect the lake sensitivity results discussed above. For instance, lake evaporation was estimated by the Thorthnwait Method, whose sensitivity to air temperature is relatively higher than in the Penman Method (Yin and Nicholson, 1998). Besides, the lake size and morphology are important factors affecting the lake's sensitivity to climatic variability (Olaka *et al*., 2010).

It is also important to note that the sensitivity analysis done through the linear model demonstrated a negative correlation between lake level changes and evaporation in the catchment of the lake (Figure 4.9; Table 4.8). The model was less powerful with no significant relationship between the two variables probably due to lack of consistency in water level data and the short duration of evaporation data collection.

Currently (2020-2021), the losses of the water from the lake and its tributaries due to abstraction for irrigations and evaporation in the lake catchment have decreased by 69% and 37%, respectively, making the lake storage capacity, to roughly increase by about 75% from the period 1970-1995 to the period of January 2008-June 2021. This increased storage may explain the flooding of the banks of the lake over the last two years thereby expanding the lake surface area and volume (Welcomme, 2008; Grownaris *et al*., 2015).

The results of the water balance model of Lake Baringo showed that precipitation and runoff represent 25% and 75% in 1970-1995 and 29.4% and 70.6% in 2020-2021 of the overall inputs, respectively (Table 4.9) indicating that the lake is mainly fed by its tributeries than the direct rainfall water in the basin. These results are somehow in disagreement with estimates in Lake Kivu (54 vs. 46 %) of Bergonzini (1998) and (55 vs. 45 %) of Muvundja *et al*. (2014), Lake Victoria (Tate *et al*., 2004; Sewagudde, 2009) and Lake Tana (Kabede *et al*., 2006) where the precipitation is the highest contributor (> 50%) of the overall inputs. The results, therefore, imply a greater management of the lakes watershed in order to ensure continued supply of water to the lake. Evaporation and other losses (abstractions and seepage), represented 59.3 and 36.0 % in 1970-1995 and 41.8% and 11.1 % in 2020-21 (Table 4.9) of the total water losses, respectively, indicating a high lake level sensitivity to meteorological variable changes (Russell and Johnson, 2006). These results are in agreement with the results from other African lakes where evaporation is the main source of lake water losses (Bergonzini, 1998; Kabede *et al*., 2006; Kumambala and Ervine, 2010; Muvundja *et al*., 2014; Mbanguka *et al.*, 2016).

5.3 Effects of lake level fluctuations on water quality variables and fisheries

The results of this study highlighted inter- and intra-annual water level variations in Lake Baringo and clear patterns across years for the long-term water level database (1956-2020). They also demonstrated a nearly two-decade periodicity or oscillation in extreme peak water levels across a 64-year time frame. This periodicity is likely associated with oscillation in abnormal rainfall events in the region due to climate cycle variability in the lake watershed over time (Ngaira, 2006; Aura *et al.,* 2020) likely caused by interannual changes of the Indian Ocean dipole (Bergonzini *et al*., 2004; Thiery *et al*., 2014, 2015; Hirons and Turner, 2018).

The average annual water levels (WLs) and amplitude intensities have been fluctuating in Lake Baringo over the last six decades, as also witnessed in other East-African lakes (Bergonzini, 1998, 2004; Muvundja *et al*., 2014; Kolding and Van Zwieten, 2012; Hirons and Turner, 2018) and elsewhere (Blenckner, 2005; Neckles, *et al*., 1990; White *et al.*, 2008). The annual level changes in Lake Baringo are characterized by variable water quality changes ranging from poor to good water quality months and years with likely resultant effects on lake ecology (Hickley *et al*., 2004) and livelihoods (Aura *et al*., 2020). In the long-term dataset, the lake level has fluctuated from a low of 1.47 m in 1956 to peak levels (13.95 m) in 2017. The findings show Lake Baringo water levels to be unstable as evidenced in several African lakes (Lakes Turkana, Malawi, Kivu, Victoria, Tanganyika, and others) since the early 1960s due to severe drought and flood events (Mercier *et al*., 2002; Boko *et al*., 2007, Muvundja *et al.*, 2014). These lake level variations have been related to the warning over the Indian Ocean during the years of *El Nino*–southern oscillation (ENSO) event (Mercier *et al*., 2002; Bergonzini *et al.*, 2004; Thiery *et al*., 2014a, 2015; Hirons and Turner, 2018) leading to flooding in the East African Lakes region. The years with lower water levels in Lake Baringo and others are attributable to probably drought situations likely caused by *La Niña* events induced by anomalies of low zonal wind intensity (ZWI) over the Indian Ocean (Bergonzini *et al*., 2004; Khaki, and Awange, 2021).

The mean monthly lake level dramatically increased during wet season from April 2020 to November 2020, while during the dry months (December 2020 to June 2021), the lake level progressively dropped down indicating that the precipitation plays a key role in the lake level changes in Lake Baringo basin. These results suggest that the rain to be the potential driver of the seasonal shift of the lake water quality and trophic state as observed elsewhere (González-Bergonzoni *et al.*, 2016). At monthly scale, the results also indicated that the very dry months of January and February for both years 2020 and 2021 provided a good water quality of the lake as WQI approached the WHO recommended value of WQI ≤ 100 indicating less influence of nutrients loads with associated particules and erosion effets from the lake basin through rivers. This is related to the location of the Lake Baringo basin in a semi-arid region with less precipitation (600 mm and 900 mm) but high annual evaporation rate of 1650-2300 mm (Odada *et al*., 2006) and water abstraction from rivers upstreams for agricultural purposes (Nyakeya *et al*., 2020, Walumona, Pers. Obs.). The lake depends mostly on its tributaries originating from the humid and hilly areas where the annual precipitation is very high varing between 1,100 mm and 2,700 mm (Odada *et al*., 2006). These results also indicated that in general, the lake level is unstable and that the water of Lake Baringo was of good quality in months closest to the highest long-term monthly mean (DLTM = 1.3 m), and of bad quality in months with decreasing lake levels indicating the role of dilution on the water quality of the lake (González-Bergonzoni *et al.*, 2016).

In addition to natural events (flood and droughts), human watershed activities such as irrigated agriculture, damming of rivers inflowing to the lake, livestock, and domestic abstractions may have contributed to variations in lake levels through influence on water balance components (Odaada *et al*., 2006; Nyakeya *et al*., 2020; Herrnegger *et al*., 2021). In this study, a significant direct correlation between long-term fisheries production, fish species average condition, and water level changes was found indicating that the WLFs are the main driver of fisheries production and water quality variation in the lake. The fisheries production-lake level relationship has also been reported for Lake Turkana in Kenya (Kolding, 1993a) and elsewhere (Gasith and Gafny, 1990; Neckles *et al*., 1990; Gasith *et al.*, 1996; Gafny *et al*., 1992; Kolding *et al*., 2012) and may be due to mechanisms related to water-level mediated changes in fish species catchability. The concordance of water quality variables with water level fluctuations (WLFs) is expected as water levels are directly controlled by hydrological inputs driven by severe droughts and floods in the region. An increase or draw-down in water level causes a shift in the lake's hydrological budget that might affect the ecological process in the lake (Gafny *et al*., 1992; Canning *et al*., 2017). The relationship between water level (WL) and the lake surface area demonstrated that a 1 m increase in water level leads to about 90 km^2 change in Lake Baringo’s surface area. This finding indicated that high water levels create flood pulse regimes that provide enhanced littoral habitats, food, and breeding areas for fish species that contribute to increased fisheries yields (Gownaris *et al*., 2015).

Increased inter- and intra-annual water level fluctuations and the associated interactions between aquatic and terrestrial ecosystems allow the nutrient pump to boost primary production among other effects (Karenge and Kolding, 1995; Kolding and van Zwieten, 2012). Studies in African lakes (Lakes Kariba in Zambia/Zimbabwe and Turkana in Kenya) and elsewhere, demonstrated significant concordances between fisheries production, and water level fluctuations (Melack, 1976; Downing *et al.*, 1990; Kolding and van Zwieten, 2012). In Lake Baringo, the lake level decline leads to significant losses of the open water habitat, thereby reducing the carrying capacity of species that dominate its pelagic zone. Moreover, the lake level decline shrinks the floodplain area and leads to losses of littoral habitats, feeding and breeding habitats of some species (Karenge and Kolding, 1995; Mageria and Kibwage, 2009;

Grownaris *et al*., 2015). This might probably be the reason for the low catchability of some commercially valuable fishery species (e.g. *O. niloticus baringoensis* and *Labeobarbus intermedius*) of Lake Baringo during the years of lower levels in the lake (Mlewa *et al*., 2005; Nyakeya *et al*., 2020). In Lake Turkana, Kenya, the majority of the lake's endemic species are found below the 10-m contour indicating that declines in inshore and offshore habitats would have severe ecological consequences for these species (Hopson, 1982; Grownaris *et al*., 2015).

Regression and Gaussian analysis outputs in this study demonstrated the highly responsive nature of turbidity and WQI to increasing DLTM, indicating that hydrological indices are the major drivers of the water quality changes in the lake. The results of the influence of lake level on physico-chemical variables are in agreement with the findings from most tropical reservoirs and lakes (Winfried, 2004; Wulandari, 2015). For example, in Lake Tana, Ethiopia, the plant nutrients are quickly exhausted during draw-downs, and their effects on biological production become less important (Karengea and Kolding, 1995). The results of this study also indicated that the periods of low WLs were characterized by lower lake primary production measured as chlorophyll-a concentrations, indicating a negative effect of WLFs on the lake production in terms of phytoplankton biomass in the short and long-terms. This would also likely explain the reported secondary production (zooplankton) limitation in Lake Baringo (Schagerl and Oduor, 2003; Tarras-Wahlberg *et al*., 2003). The observed changes in water quality parameters and fisheries production as a result of lake level changes emphasizes the need for long-term monitoring of the lake's catchment and the need for an integrated lake management approach. There is also a need of implementing appropriate climate adaptation and mitigation measures in the lake catchment area in order to mitigate the influence of lake level variability.

5.4 Modeling the food web properties and fisheries dynamics in Lake Baringo using Ecopath mass-balanced model

5.4.1 Trophic models of Lake Baringo

The three trophic models of the lake all indicated lowed top predation abundance. In the 1999 lake model, there is low predator and zooplankton biomass with subsequent increase in phytoplankton biomass. In the 2010 and 2020 models, there is increased zooplankton abundance perhas as a result of lower zooplanktivore abundance but this was not enough to depress phytoplankton biomass significantly. The 2020 model of the lake shows a greater detrital content in the food web. Based on the findings of Pauly *et al.* (1998), the 2010 trophic model of Lake Baringo indicated less biomass of zooplanktivore (Table 4.13) resulting in an increase in phytoplankton biomass (the food of *O. niloticus baringoen sis*) due to the overexploitation of zooplanktons. This trophic cscade could explain the rareness of *O. niloticus* in the fishermen's catches during the same period among other reasons (Nyakeya *et al.*, 2020).

The ecotrophic efficiency (EE) values of detritus and of phytoplankton in the lake were fluctuating over time indicating variation in relative importance of the resource in the system over time. The instability of detritus and phytoplanktons in the lake maybe also related to the fluctuating water quality of the lake due to the changing trophic state of the lake caused by the anthropogenic activities in the catchment (Walumona *et al*., 2021). For phytoplankton, the EE values show increase in its utilization in the lake over time by its consumers’ (mainly zooplankton and *O.*

niloticus) affecting negatively their biomasses. Also, the findings of the three models indicate increases in the biomasses of the fish functional groups over time suggesting a recovery process of the lake fishery (González-Bergonzoni *et al.*, 2016) also supported by increasing trend in the lake's overhead (ability to recover) (Table 4.15). Lower *EE* values for primary producers (phytoplankton and detritus) indicating the food availability in the ecosystems have been found in Lakes Volta (Mensah *et al*., 2019), Awassa (Fetahi and Mengistou, 2007), and in Manwan Reservoir (Deng *et al*., 2015).

The *EE* value for the top predator (*Protopterus aethiopicus*) in the Lake Baringo ecosystem including other commercial fish species (*C. gariepinus, O. niloticus* and *L. intermedius*) was zero (0.000) for all the trophic models, indicating their low predation rate and that their stocks have been underexploited due to predaction. This is likely due to lack of secondary predators in the lake or maybe a data weakness as evidenced by low pedigree values.

The decline in the cycling rates (Finn's index) and mean transfer efficiencies (TE) from 1999 to 2020 suggests the lake to be less efficient in inorganic recycling and more susceptible to environmental disturbances over time (Finn, 1976). These findings are typical of systems that are ecologically (Odum, 1969). Based on cycling rates (Finn's index and inflow indices (system omnvory, connectance index; CI), and others that the lake ecosystem could be categorized as immature, has been recovering and, becoming progressively resistant against perturbations over time (Strayer, 1991). The decrease in Finn's cycling Index (FCI) throughout the period of the study especially between 2010 and 2020 related probably to the lower abundance of filter feeders and other recycling agants in the lake ecosystem compared to other exploited ecosystems (Ortiz *et al*., 2015). The total production for Lake Baringo (2010 and 2020) and Lake Kariba was much higher, compared to other African lakes such as Lakes Awassa, Volta, Malawi and Victoria. This suggests high productivity in Lakes Baringo and Kariba compared to others which gives scope for more manipulations in the lake (Hickley *et al.*, 2004).

5.4.2 Ecological functioning indicators

Total system throughput (TST) is the sum of all flows in the ecosystem: sum of all consumption, the sum of all exports from the ecosystem, the sum of all the respiration flows and the sum of all flows into the detritus representing the size of the entire ecosystem (Ulanowicz, 1986). The TST values of Lake Baringo (Table 4.15) are high for the three ecopath models emphasizing the ecological immaturity stage of the lake related to its young geological age (Odum, 1969). Other studies on tropical ecosystems (Abdul and Adekoya, 2016) have been reported to significantly have high TST values indicative of young relative age (Polovina, 1984). The net system production (NSP) of Lake Baringo varied between 964.46 t km^2 yr^{-1} in 2010 and 5,950.99 t km^2 yr^{-1} in 1999 supporting the notion that the lake's ecosystem is in an immature developmental stage as suggested by Kumar *et al*. (2015).

The TPP/TR and TPP/B ratios (Table 4.16) are indicators of the degree of system maturity (Odum, 1969; Christensen, 1995). At the young stage of maturity, the ratio TPP/TR is greater than 1, the primary production exceeding the rate of community respiration (Odum, 1969). The ratio between the total primary production and total biomass, (TPP/TB) and the ratio between total primary production and total respiration (TPP/TR) showed high values for all the three trophic models (1999, 2010, 2020) suggesting the immaturity of the lake's ecosystem. At the maturity stage, ecosystems are characterized by lower values of TPP/B ratio related to biomass accumulates as the ecosystem develops (Odum, 1971). The system omnivory Index (SOI) in Lake Baringo was lower over time highlighting the low diversity in diets and lack of specialists also characteristic of developing systems (Kumar *et al*., 2015). The recent geological origin of Lake Baringo supports Odum's theory (Odum, 1969) that ecosystems at the youngest stages of maturity are characterized by higher ration TPP/TR as found for Lake Baringo.

In general, the fishery indicators (TLc, PPR and GFE) showed lower and fluctuating values indicating that the lake's fishery has been less sustainable characterized by the changing fishing intensity based mostly on the apex predator (*P. aethiopicus*) over time. These results indicate the possibility of Lake Baringo's ecosystem to be affected by both ecosystem dynamics (such as predator-prey interactions) (Angelini *et al*., 2006) due probably to the low diversity in diets and lack of specialists and external factors (such as intensity of fishing) over time (Heymans *et al*., 2004; Shannon *et al*., 2004; Libralato, 2008).

Keystone functional groups are defined as species with a structuring role within ecosystems and the food webs that interconnect with a relatively low biomass and hence food intake (Power *et al*., 1996). They strongly also influence the abundances of other species and the ecosystem dynamics (Piraino *et al*., 2002). The index assigned relatively low keystoneness to functional groups with high abundance (e.g., *P. aethiopicus* and *C. gariepinus*) and high keystoneness to functional groups with low abundance (*O. niloticus* and zooplankton). The model identified *O. niloticus baringoensis* to be the keystone species in Lake Baringo affecting the population structure in the lake.

Another important result is that *Oreochrimis niloticus* affects the population structure in the lake through top-down influences characteristic of keystone species (Paine, 1969; Davic, 2003)). This result concurs with those of Libralato *et al.* (2006) where keystone species did not always exert their high impact by means of top-down effects. However, results found in many ecosystems indicated that keystone functional groups exert their effect via top-down controls (Wootton, 1994; Wootton *et al.*, 1996; Berlow, 1999). Nonetheless, bottom-up influences can also be important as shown in this study for *O. niloticus baringoensis* and elsewhere (Bustamante *et al*., 1995; Menge, 1995; Libralato *et al.*, 2006) without contradicting previous findings that demonstrated the high importance of top-down controls in keystoneness (Paine, 1966; Menge *et al*., 1994; Estes *et al*., 1998).

The three trophic models (1999, 2010, 2020) calculated lower pedigree index values suggesting some likely uncertainties in the input data due to probably the lack of certain scientific information on some key functional groups such as benthic communities, macrophyte, mammals (hyppos and crocodiles), not used in the model, and to certain gaps in the fisheries data used in the ecopath model of Lake Baringo. The slight difference in the Pedigree index among the three models (1999, 2010, 2020) can be related to the different data sources and the need to re-parameterize the model when more data become available. According to Villanueva, *et al.* (2006), the estimation of data inputs in the model from *in situ* sampling wheneve possible is recommended to avoid uncertainties associated with data collection from lilerature and/or similar models. The Ecopath models developed in this study are pioneering for the lake and most lakes in Kenya, and constitute the first and useful trophic tool to explain the lake's functioning. These results are also useful for fisheries management decision makers in understanding temporal trends in lake processes and fisheries. However, the novel will require reparametization in future when more data becomes available as is the case for many pioneering models (Dowling *et al*., 2012; Natugonza *et al*., 2016).

CHAPTER SIX

CONCLUSIONS AND RECOMMENDATIONS

6.1 Conclusions

This study evaluated the water quality, fisheries and trophic status of Lake Baringo, a rift valley lake in Kenya designated as a Ramsar site due to its high biodiversity. Additionally, the lake's water balance components were modeled and its sensitivity to meteorological variables was evaluated. Monthly WQI values exceeded 100, the WHO upper limit for drinking water indicating that the lake's water is not of good quality (unsuitable) for human consumption. The poor quality of water related to high value of WQI in the lake was attributed to the higher values of turbidity caused by rainfall-mediated erosion in the catchment. There is a need for a comprehensive and integrated lake catchment management plan to combat the soil erosion in the Lake Baringo watershed. The organic pollution index (OPI) results showed that the Lake water is not organically polluted; however, the spatial variation in TN and TP concentrations suggests the possibility of localized eutrophication. Nutrients with the highest relative weights (TP, TN, NO_3^-, NO_2^-, NH_4^+, PO_4^{3-}) showed less effect on water quality indices (WQI and OPI) indicating minimal effects of agricultural runoff on the lake. The lake's trophic status has been changing over time from hypereutrophic to mesotrophic indicating temporal variability in anthropogenic influences.

This study is the first attempt to assess the water balance components of Lake Baringo, in order to produce a data-based information needed for better management of the water resources of small lakes in the dry areas. The lake outflows used in the model were based on the underground seepage, the evaporation rates calculated empirically, and the abstraction of water from the lake tributaries for irrigation purposes. The inflows are constituted by the rainfall estimated from meteorological stations in the lake watershed and the groundwater flux. The results showed that for both 1970-1995 and 2008-2021 periods, the total inflows into the lake is dominated by the surface runoff contributions of 75% and 71%, respectively, while the lake loses most of its water through evaporation; 59% during 1970-1995 and 42% during 2008-June 2021. A 13% reduction in the annual water abstraction was observed from 1970 to 2021with a rise of 9% of evaporation rates resulting in a 75% increase of the water storage in the lake. This contributed to the maintenance of the stability of the water storage in the lake. The results of lake level sensitivity analysis to climatic variability showed the lake to be sensitive to rainfall and Evaporation indicating that the decline in rainfall or the rise in Evaporation from their current conditions woud lead to significant drop in water levels in Lake Baringo. The results of water balance analysis are useful for an integrated approach to the lake's basin management that includes land-use controls. A temperature decrease of about 2°C is projected to lead to lake's overflow.

The results also highlight the link between water quality properties, fishery yields, species condition, and WLFs in Lake Baringo. These linkages result from the influence of WLFs on the critical water quality parameters in this lake like turbidity, DO, electrical conductivity, temperature, and the water quality index (WQI). There is a direct link between years of high lake level and increased fisheries landings perhaps mediated through increased habitat availability as nursery and feeding grounds for species, however, the precise mechanisms will require further investigations. During periods of droughts with less inflow due to both natural (climate variability) and anthropogenic influences (damming of inflow rivers for water abstraction for both irrigation and drinking purposes), the water level decreases in tropical lakes, like Lake Baringo, which may lead to a decline in littoral habitats due to the water volume reduction, and to changes in ecological functions. For example, species may be predisposed to high predation pressure due to habitat shifts and reduced refugia areas. The severe decline in lake levels (case of Lake Baringo) may significantly alter the physico-chemical characteristics of African lakes with resultant effects on the ecological structure and functioning of these lakes. The observed changes in water quality parameters and fisheries production as a result of Lake Baringo level changes emphasizes the need for long-term monitoring of the lake's catchment and the need for an integrated lake management approach.

The present study is the first completed Ecopath model analyzing the trophic aquatic ecosystems structure dynamics and ecosystem functioning for Lake Baringo, and for most lakes in Kenya. The structure of the food web in Lake Baringo comprised three trophic levels in the three annual models, with *P. aethiopicus* species feeding at the highest trophic level and *O. niloticus baringoensis* the keystone fish species in the lake. Network analysis demonstrated the importance of detritus and phytoplankton groups in the food web (trophic level I) and their susceptibility to predation changes by fish functional groups at higher trophic levels in the lake. The lower EE values at the trophic level I indicate less demand of primary producers (detritus and phytoplankton) suggesting a weak bottom-up control in the lake's ecosystem. The network analysis using ecosystem maturity indices (TPP/TR and TPP/B) described

Lake Baringo as being in an unstable stage and a developing stage process towards its ecological maturity or equilibrium.

Keystoneness as calculated from EwE for the functional groups of the three (1999, 2010, 2020) food webs showed that the *Oreochromis niloticus baringoensis* (the endemic tilatipia) was a fish species characterized by high keystoneness index (KS*i*) and total relative impact (ε_i) with low biomass in the lake compared to other functional groups. Also all the fish species in the ecopath models were characterized by low EE indicating their underexploitation in the lake and/or the need of using predators such as mammals (eg. crocodiles) in the Ecopath models of the lake in order to enhance trophic cascades. There is need for more data in order to re-parameterize the generated Ecopath model of Lake Baringo for a more robust undanstanding of the lake functions.

6.2 Recommendations

The following recommendations are made following this study:

1. The findings of the present study indicate that the water of Lake Baringo, during the study periods, was of bad quality for human consumption requiring a pre-treatment policy (e.g., boiling of water, chlorination, etc.) before human consumption, and that the water quality of the lake based on the Carlson's Trophic State Index (TSI) has been fluctuating from hypereutrophic (water of very bad quality) to mesotrophic state (water of moderate quality) over time at intra- and inter-annual scales. There is need for implementation of water and land-use management policies at the watershed scale for the lake's water

quality management in order to support livelihoods and sustain ecological processes.

2. The results of this study indicate the necessity to better monitor the eco-hydrology of Lake Baringo and its catchment. There is need for controls on the water abstractions from the inflows and consequent effects on lake water levels. There is need for continuous monitoring and generation of quality data on precipitation and other meteorological variables (eg. evaporation, air temperature) on the lake and in the watershed, in addition to the measurement of discharge of rivers to the lake. This information is necessary for validating and updating the water balance components of the lake as estimated in this study. Water balance models are helpful in quantification and predictions of potential impacts of climate change and human-induced influence on lake functioning. The results indicate the need of bathymetric and topographic surveys of the lake to obtain information about temporal changes in the southern zone of the lake, the shallowest and fishing zone, as well as to obtain lake level induced shoreline changes throughout the lake for sustainable fisheries management in the lake.

3. The results of effects of WLFs on the fisheries of Lake Baringo indicate significant relationships between the average condition factor of the native species, *Oreochromis niloticus baringoensis,* and the annual lake level amplitude as well as between the water quality variables and WLFs. However, different relationships were observed between catches of the lungfish, *Protopterus aethiopicus,* and *Labeobarbus intermedius* with WLFs in the lake

indicating a species-specific influence of WLFs on species. The results demonstrate that WLFs of Lake Baringo are a significant driver of fish species biomass and physico-chemical properties of the lake. Therefore, the study emphasizes the need for long-term monitoring of the lake's limnological characteristics, and fishery production and catchment activities for the purposes of integrated lake management. Such management will require monitoring of upstream developments in order to maintain natural inflows during draw-down seasons for Lake Baringo's ecological functioning and fishery production. The results also indicated that the periods of low WLs were characterized by lower lake primary production, measured as chlorophyll-a concentrations, indicating a negative effect of WLFs on the lake production in terms of phytoplankton biomass likely affecting the biomass and diversity of zooplankton community in the lake. There is also a need of implementing appropriate climate adaptation and mitigation measures (eg. reforestation) in the lake catchment area in order to mitigate the influence of climate change and human influence (river damming) on the lake level variation for the sustainable fisheries management and the maintenance of ecological functioning of the lake.

4. The results of the Ecopath modeling suggest Lake Baringo to be in a fluctuating ecological stage with a trend towards the ecological equilibrium (maturity). The models showed that the ecotrophic efficient (EE) was equal to zero for all fish species suggesting their underexploitation in the lake (relative to optimum offtakes) or the absence of their consumers in the models. This may be due probably to the lack of scientific data on all the predators in the lake and may affect the accuracy of the model. The fisheries indicators (PPR, TLc and GFE) and the keystoneness index (KSi) indicated that Lake Baringo's fish biomass also fluctuated over time due probably to the lake level changes, and that *O. niloticus baringoensis* functions as a keystone species in the lake. Fisheries management should ensure sustainable catch of *O. n. baringoensis* in order to preserve its keystone function in the lake. Model outputs indicate the lake's ecological resilience capacity has increased over time but the contributing factors are not clear. The generated Ecopath model of the lake will require re-parameterization in the future when more data becomes available on the functional groups not included in the model.

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APPENDICES

Appendix I: Identification Keys for phytoplankton population: a large file available online:

1.http://www.kaowarsom.be/documents/MEMOIRES_VERHANDELINGEN/Sciences_naturelles_medicales/Nat.Sc.(NS)_T.23,2_MPAWENAYO,%20B._Les%20eaux%20de%20la%20plaine%20de%20la%20Rusizi%20(Burundi)-%20les%20milieux,%20la%20flore%20et%20la%20v%C3%A9g%C3%A9tation%20algales_1996.PDF

2. http://nio.org/userfiles/file/biology/Phytoplankton-manual.pdf

3. http://oceandatacenter.ucsc.edu/home/outreach/PhytoID_fullset.pdf.

Appendix II: Supplementary Tables (S)

Table S1: Two-way ANOVA (P < 0.05) results for 10 selected variables measured in Lake Baringo, Kenya, for the period of 2008-2020. Bold figures depict parameters with no significant differences of the interaction between stations x years

Variables	*Source of variations*	*SS*	*DF*	*MS*	*F*	*p_value*	*F crit*
a) D0	Stations	72.24	4	18.06	31.73	0.000	2.38
	Years	118.31	12	9.86	17.32	0.000	1.77
	Interaction	48.68	48	1.01	1.78	0.001	1.38
	Within	406.99	715	0.57			
	Total	646.21	779				
b) Temperature	Stations	2304.27	4	576.07	5.78	0.000	2.38
	Years	2556.11	12	213.01	2.14	0.013	1.77
	Interaction	9528.56	48	198.51	1.99	0.000	1.38
	Within	71244.65	715	99.64			
	Total	85633.60	779				
c) Conductivity	stations	203942.68	4	50985.67	8.47	0.000	2.38
	Years	8880468.06	12	740039.00	122.99	0.000	1.77
	Interaction	865857.23	48	18038.69	3.00	0.000	1.38
	Within	4302358.68	715	6017.28			
	Total	14252626.66	779				
d) Turbidity	Stations	5336.12	4	1334.03	1.21	**0.306**	2.38
	Years	1004803.33	12	83733.61	75.80	0.000	1.77
	Interaction	47144.46	48	982.18	0.89	**0.686**	1.38
	Within	789832.88	715	1104.66			
	Total	1847116.80	779				
e) Secchi disk	Stations	7172.74	4	1793.19	7.89	0.000	2.38
	Years	915122.45	12	76260.20	335.74	0.000	1.77
	Interaction	12323.74	48	256.74	1.13	**0.257**	1.38
	Within	162405.87	715	227.14			
	Total	1097024.81	779				
f) Chlorophyll-a	Stations	10396.76	4	2599.19	2.92	0.022	2.41
	Years	125119.62	12	10426.63	11.72	0.000	1.79
	Interaction	51224.26	48	1067.17	1.20	**0.188**	1.41
	Within	231358.88	260	889.84			
	Total	418099.52	324				

g) TP	Stations	125742.41	4	31435.60	34.53	0.000	2.41
	Years	1221866.16	12	101822.18	111.86	0.000	1.79
	Interaction	306163.42	48	6378.40	7.01	0.000	1.41
	Within	236666.48	260	910.26			
	Total	1890438.47	324				
h) TN	Stations	82870.02	4	20717.50	3.41	0.010	2.41
	Years	7701595.85	12	641799.65	105.64	0.000	1.79
	Interaction	188182.01	48	3920.46	0.65	**0.966**	1.41
	Within	1579617.06	260	6075.45			
	Total	9552264.93	324				
i) SiO_4^{4-}	Stations	1803.76	4	450.94	26.37	0.000	2.41
	Years	3247.06	12	270.59	15.82	0.000	1.79
	Interaction	2850.20	48	59.38	3.47	0.000	1.41
	Within	4446.47	260	17.10			
	Total	12347.49	324				
j) PO_4^{3-}	Stations	9078.65	4	2269.66	21.15	0.000	2.41
	Years	21711.70	12	1809.31	16.86	0.000	1.79
	Interaction	22427.61	48	467.24	4.35	0.000	1.41
	Within	27905.86	260	107.33			
	Total	81123.82	324				

DO: Dissolved oxygen, TP: total phosphorus and TN: total nitrogen, SiO_4^{4-}: silicates and PO_4^{3-} : phosphates

Table S2: Two-way ANOVA (P < 0.05) results for 15 selected variables measured in Lake Baringo, Kenya, for the period of January 2020- June 2021. Bold figures depict parameters with significant differences of either both the seasons and stations or one of them, theinteraction between station x years did not show significant differences between the parameters

Variables	*Source of variations*	*SS*	*DF*	*MS*	*F*	*p_value*	*F crit*
a) DO	Seasons	5.70	1	5.70	4.38	**0.0382**	3.907
	Stations	40.36	8	5.05	3.87	**0.0004**	2.003
	Interaction	9.15	8	1.14	0.88	0.537	2.003
	Within	187.67	144	1.30			
	Total	242.88	161				
b) Temperature	Season	1.50	1	1.50	0.49	0.484	3.907
	Stations	43.63	8	5.45	1.78	0.086	2.003
	Interaction	20.17	8	2.52	0.82	0.584	2.003
	Within	441.71	144	3.07			
	Total	507.007	161				
c) Conductivity	Season	3508.52	1	3508.52	0.56	0.46	3.91
	Stations	74721.87	8	9340.23	1.49	0.16	2.00
	Interaction	50908.02	8	6363.50	1.02	0.43	2.00
	Within	900265.29	144	6251.84			
	Total	1029403.7	161				
d) SD	Season	4142.52	1	4142.52	12.64	**0.0005**	3.91
	Stations	5853.06	8	731.63	2.23	**0.028**	2.00
	Interaction	351.59	8	43.95	0.13	0.998	2.00
	Within	47209.16	144	327.84			
	Total	57556.33	161				
e) Chl-a	Season	75.66	1	75.66	0.02	0.88	3.91
	Stations	39314.98	8	4914.37	1.52	0.16	2.00
	Interaction	24424.99	8	3053.12	0.94	0.48	2.00
	Within	466713.81	144	3241.07			
	Total	530529.44	161				
f) TP	Season	2347.61	1	2347.61	0.76	0.38	3.91
	Stations	37392.19	8	4674.02	1.52	0.15	2.00
	Interaction	11948.84	8	1493.61	0.49	0.86	2.00
	Within	442309.89	144	3071.59			
	Total	493998.53	161				
g) TN	Season	14297,03	1	14297.03	13.631	**0.0003**	3.91
	Stations	16839,10	8	2104.89	2.007	**0.0495**	2.00
	Interaction	5359,46	8	669.93	0.639	0.7441	2.00

	Within	151033,4	144	1048.84			
	Total	187528,99	161				
h) SiO_4^{4-}	Season	113.14	1	113.14	1.36	0.246	3.91
	Stations	1430.10	8	178.76	2.15	**0.035**	2.00
	Interaction	239.95	8	29.99	0.36	0.939	2.00
	Within	11988.62	144	83.25			
	Total	13771.81	161				
i) NO_3^-	Season	605.88	1	605.88	1.26	0.263	3.91
	Stations	5897.13	8	737.14	1.54	0.149	2.00
	Interaction	6434.74	8	804.35	1.68	0.108	2.00
	Within	69006.89	144	479.21			
	Total	81944.64	161				
j) NO_2^-	Season	1.84	1	1.843	0.043	0.836	3.91
	Stations	716.05	8	89.507	2.077	**0.041**	2.00
	Interaction	113.43	8	14.178	0.329	0.954	2.00
	Within	6206.59	144	43.101			
	Total	7037.92	161				
k) PO_4^{3-}	Season	2843.68	1	2843,68	4.25	**0.0411**	3.91
	Stations	6447.56	8	805.94	1.20	0.301	2.00
	Interaction	5032.98	8	629.12	0.94	0.486	2.00
	Within	96437.10	144	669.70			
	Total	110761.32	161				
l) NH_4^+	Season	12.99	1	12.99	0.008	0.927	3.91
	Stations	16006.73	8	2000.84	1.289	0.253	2.00
	Interaction	9162.92	8	1145.37	0.738	0.658	2.00
	Within	223445.02	144	1551.70			
	Total	248627.66	161				
m) pH	Season	5.457	1	5.457	14.919	**0.00016**	3.91
	Stations	3.112	8	0.389	1.064	0.392	2.00
	Interaction	1.272	8	0.159	0.435	0.899	2.00
	Within	52.668	144	0.366			
	Total	62.510	161				
o) TDS	Season	13470.332	1	13470.332	9.652	**0.002**	3.91
	Stations	46282.214	8	5785.277	4.145	**0.00017**	2.00
	Interaction	18103.050	8	2262.881	1.621	0.123	2.00
	Within	200971.576	144	1395.636			
	Total	278827.171	161				
p) WQI	Season	68689.278	1	68689.278	6.759	**0.010**	3.91
	Stations	170866.651	8	21358.331	2.102	**0.039**	2.00
	Interaction	146309.228	8	18288.654	1.799	0.081	2.00
	Within	1463331.421	144	10162.024			
	Total	1849196.582	161				

Table S3: Annual catch landings, surface water temperature, length and growth coefficient estimates of fish species used in Ecopath model for Lake Baringo (Kenya) for 1999, 2000, and 2020

	Species name/group	Catches (t/year)	*%*	*n*	*Lc (cm)*	*Lmax (cm)*	*Lmean (cm)*	*L∞* (cm)	K	T_c (°C)
Periods (Years) 1999										
1	*Protopterus aethiopicus*	42.1	72.2	453	32	136.0	86.5	133.4	0.40	26.99
2	*Labeobarbus intermedius*	0.1	0.2	259	16	40.5	27.2	42.0	0.94	26.99
3	*Clarias gariepinus*	12.8	21.9	458	18	180.0	52.3	186.9	0.32	26.99
4	*0reochromis niloticus*	3.3	5.7	335	10	33.0	21.9	33.6	1.20	26.99
	Total	**58.3**	**100**	**1,505**						
Periods (Years) 2010										
1	*Protopterus aethiopicus*	38.3	54.9	509	21	148.0	81.0	150.7	0.33	27.40
2	*Labeobarbus intermedius*	0.5	0.7	633	15	70.3	25.1	75.1	0.48	27.40
3	*Clarias gariepinus*	12.92	18.5	631	18	141.0	48.3	147.0	0.27	27.40
4	*0reochromis niloticus*	18.04	25.9	652	10	62.5	22.2	64.1	0.67	27.40
	Total	**69.76**	**100**	**2,425**						
Periods (Years) 2020										
1	*Protopterus aethiopicus*	42.4	37.1	5376	10	160.0	75.5	168.0	0.25	26.78
2	*Labeobarbus intermedius*	9.8	8.6	5889	13	100.0	23.0	108.2	0.01	26.78
3	*Clarias gariepinus*	33.6	29.4	2016	17	102.0	44.3	107.1	0.22	26.78
4	*0reochromis niloticus*	28.5	24.9	1027	10	92.0	22.6	94.5	0.14	26.78
	Total	**114.3**	**100**	**14,308**						

Abbreviations: Lc: length at first capture; Tc: mean annual surface water temperature; *L*max: maximum length or total length; *Lmean*: mean length; *L*∞: asymptotic length; *k*: growth coefficient and n stands for the number of fish specimens.

Appendix III: Similarity Report

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