Addis Ababa University

College of Natural and Computational Science

School of Graduate Studies

Department of Zoological Sciences

Assessment of water quality, phytoplankton, zooplankton and trophic interaction of lake Tinshu abaya

In Partial Fulfillment of the Requirement for Master of Aquatic Ecosystem and Environmental Management (AEEM)

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Addis Ababa, Ethiopia

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Approved by the Examining Committee

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Declaration

I, the undersigned, declare that this thesis was written entirely by myself and that it has not previously been presented, in whole or in part, in any application for a degree or other qualification. Except as referenced or acknowledged, the work provided in this thesis is completely mine.

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List of abbreviations

a.s.l.	Above sea level
ANOVA	Analysis of Variance
APHA	American Public Health Association
Chl-a	Chlorophyll-a
DO	Dissolved oxygen
EBM	Ecosystem based management
EE	Eco trophic efficiency
RDA	Redundancy Analysis
SD	Standard deviation
TDS	Total Dissolved Solids
TSS	Total Suspended Solids
TSS	Total Suspended solids
Zeu	Euphotic depth
ZSD	Secchi depth

Abstract

Freshwater ecosystems are subjected to various anthropogenic activities, which result water quality degradation and ultimately affect the overall functioning of ecosystem. Therefore, *p* monitoring and evaluating fresh water ecosystems limnological and biological aspect is crucial. To design sustainable resource the present aimed to assess some physico-chemical and biological water quality parameters, the trophic structure and energy flow of the lake. Samples were taken monthly from three sites from May to August 2022. All the data samples were collected and analyzed using standard methods of sampling and analysis. Composite samples were used for phytoplankton analysis, while 70µm mesh size plankton nets were used for zooplankton sampling. The trophic interactions, energy flows, and ecosystem properties of ecosystem were analyzed by ecotrophic modeling using Ecopath with Ecosim software (EwE, 6.6 version). Primary and secondary data were used to construct the mass balance model of the lake ecosystem. The results indicate that the lake had fairly high temperature (20.83_25.01 ^{o}C), and was well oxygenated (7.42_13.86 mg L^{-1}). had very turbid water (350 - 735 NTU) and shallow Secchi depth (0.082 m) and high levels of nutrients and alkalinity. The levels of all physicochemical parameters considered in the study did not differ significantly among sites (ANOVA, $p > 0.05$) although their temporal variations were statistically significant (ANOVA, $p < 0.05$). The N:P Redfield ratio indicates that the limiting nutrient in the lake phosphorus. The results of principal component analysis show the contribution of TDS, Temperature, nitrate, nitrite, ammonia and Turbidity to water quality was high, while those of SRP and SiO_2 was low. A total of 32 phytoplankton taxa belonging to 6 divisions including Bacillariophyta (Diatoms), Chlorophyta *(*Green algae), and Cyanobacteria (Blue-green algae), Euglenophyta (Euglenoids), Cryptophyta, and Dianophyta. The cyanobacteria were the most abundant during the all sampling months and followed by Bacillariophyta. Phytoplankton biomass as Chl-a ranged from 9.12 to20.08 µg L-1, with a mean value of 14.6 µg L-1. The minimum value was recorded during the rainy season (July) while the maximum was in May. Redundancy Analysis (RDA) of the relationships between environmental parameters and distribution of abundant phytoplankton showed that *Cylindrospoermopsis cuvispora* and *Cylindrospoermopsis raciborski* were correlated positively with TDS, Conductivity and turbidity. 25 typical tropical species of zooplankton were identified in samples from the lake, of which 60% species belonged to Rotifera, while 8% of the species belonged to Copepoda and 32% to Cladocera. Total zooplankton abundance was dominated by copepods followed by rotifers. Among the rotifers. *Brachionus sp., Filinia sp*, and *Keratella tropica*, were the most abundant and persistent, while *Thermocyclops decipiens* and *Mesocyclops aequatorialis* were the most abundant and persistent copepods. The biovolume of the copepod species were the highest, while that of the rotifers was the lowest. According the results of Redundancy Analyses (RDA). *T. decipiens* and *B. calciflorus* were correlated positively with conductivity and TDS. The Ecopath model identified 7 functional groups: waterfowl(birds), Nile Tilapia, carnivorous zooplankton, herbivorous zooplankton, phytoplankton, and detritus. The trophic levels of the groups varied from 1.00 for primary producers and detritus to 3.56 for predator birds. The ecosystem maturity indices total primary production/total respiration and Connectance index were 1.19 and 0.555, respectively, showing that the lake's system is at developmental stage according to Odum's theory of ecosystem development. Generally, this is the first trophic model of Lake Tinshu Abaya that has provided some useful insights into the trophic interaction and functioning of the system.

Key words: ***Eco trophic modeling, Energy flows, Food web, Trophic interaction***

1. Introduction

1.1 Background and justification

Fresh water ecosystems provide various services including, regulating services such as flood control and climate regulation, water purification and retention, provisioning services such as hydropower, fish and water, and supporting services such as nutrient cycling and production of oxygen and cultural services including recreational and aesthetic values (Grizzetti *et al.*, 2016).

However, freshwater ecosystems are highly sensitive and vulnerable to anthropogenic activities (Demirak *et al.*, 2006; Fernandes *et al.*, 2007). Many freshwater bodies around the world are being degraded due to a variety of factors, including pollution and eutrophication, which are more prevalent in developing countries (Bunn, 2016). Eutrophication is the process by which a lake becomes eutrophic due to increased nitrogen and phosphorus input, which can cause a several strange ecosystem responses(Qin *et al.*, 2013). The functioning of freshwater ecosystems is also impacted by other contaminants (heavy metals and pesticides), which impacts ecosystem services demanded by humans(Schäfer *et al.*, 2011).

It is very essential to understand the effect of anthropogenic activities factors on physical, chemical, and biological features of lakes, reservoirs, rivers, and wetlands to ensure their long-term existence with their multifold benefits and services(Søndergaard and Jeppesen, 2007). The quality of the water is monitored for various categoric purposes in many developed countries. Many of Ethiopia's major lakes are either eutrophic or hyper eutrophic(Tadesse Fetahi, 2019). However, there is often no aquatic environment monitoring program in place, despite the fact that anthropogenic impacts are growing in many developing countries including Ethiopia (Bojarczuk *et al.*, 2018).

Lake Tinishu Abaya, a rift valley lake in Ethiopia, has been the focus of research works in recent years that aimed to understand different aspects of the aquatic ecosystem (Yirga Enawgaw and Brook Lemma, 2018a). However, the interaction biotic components were not investigated. Ecosystem-based management(EBM) set the foundation for studying the interaction of the functional groups in an ecosystem and has been used as a management tool in recent years(Fletcher, 2002). EBM enables the evaluation of how fishing impacts the entire ecosystem and fisheries, through producing alternative scenarios, which can be considered when formulating strategic fisheries management plans and actions (Tomasson *et al.*, 2019).Ecopath with Ecosim (EwE) is a software used to implement and operationalize EBM.

In Ethiopia, ecosystem-based studies have been conducted in some lakes and reservoirs using EwE software(Ayalew Wondie *et al.*, 2012; Gashaw Tesfaye and Wolff, 2018; Mathewos Hailu *et al.*, 2022; Tadesse Fetahi *et al.*, 2011; Tadesse Fetahi and Seyoum Mengistou, 2007).However, our understanding and knowledge of most lakes and reservoirs using a comprehensive and holistic approach through integration of essential components is limited. Thus, Lake Tinshu Abaya (LTA) is one of these ecosystems that lacks holistic study. This study aims to determine food web structure and flows of energy among the major functional groups of Lake Tinshu Abaya. To this end, EwE software version 6.6.7 was used to explore how the major components are interacting with each other. Input data collected from the literature and primary data of the essential food web components were used. In parallel, taxonomic composition of phytoplankton and zooplankton was updated and the phytoplankton biomass and zooplankton biovolume were determined. The output of the study provides essential information to understand the lake's ecological condition and to identify management priorities to ensure the sustainable management of the lake and its surrounding ecosystem.

1.2 Statement of the problem

The Ethiopian Rift Valley Lakes' basin harbors lakes, streams, and wetlands with distinct hydrological and ecological characteristics of regional and global importance. Water qualities and trophic status of several rift valley lakes including LTA were reported(Adamneh Dagne and Habtamu Tadesse, 2021; Elias Dadebo *et al.*, 2014b; Getacher Beyene *et al.*, 2022; Lencha Semaria *et al.*, 2021; Tadesse Fetahi and Seyoum Mengistou, 2007; Zenabu Yirgu, 2011).

However, with the expansion of anthropogenic activities (population pressure) and the consequent high point and non-point sources of pollution, the water quality of the lakes has deteriorated in the recent past years (Adimasu Woldesenbet, 2015; Praveen and Mukemil, 2015; Tibebe Dessie, 2017). LTA, for instance is exposed to excessive lake-water abstraction for irrigation activities and many other purposes, and the shoreline has been modified to a great extent. Eutrophication and overfishing have also been reported for the same lake(Yirga Enawgaw and Brook Lemma, 2018b). Ecosystem models are useful in assessing the effects of various external factors on ecosystem components and aid to the underlying mechanisms that drive such changes(Townsend *et al.*, 2019).

Ecosystem-based management (EBM) is an approach of managing and conserving natural resources that considers the entire ecosystem rather than focusing on specific species. Ecosystem modeling provides information that guides EBM (Musinguzi *et al.*, 2017). Ecosystem-based ecological modeling has been implemented in Africa including Ethiopia (Ayalew Wondie *et al.*, 2012; Darwish and Younis, 1982; Gashaw Tesfaye and Wolff, 2018; Moreau, 1995; Natugonza *et al.*, 2016; Tadesse Fetahi and Seyoum Mengistou, 2007; Tadesse Fetahi *et al.*, 2011).

However, there are separate research works on various ecosystem components but no studies that have implemented ecosystem-based ecological management in Lake Tinshu Abaya. The lake lacks a unified framework for ecological information to guide management decisions. The former single species models used to make management decisions estimate the biomass of various fish species and the biotic components in the system as they are not directly affected by one another. Such analyses are unable to capture interactions between species, their prey, and predators(Link, 2010),which are important to determine the abundance structure and function of the ecosystem that limits the wide range and scope of management objectives and strategies that can be explored for future management.

Accordingly, the present study aimed to understand how Lake Tinshu Abaya functioning, the major functional groups were identified, and trophic status was assessed and energy flows were described from a holistic perspective using EwE.

The study revealed the current status of the lake and will serve as an input in the development of strategies for sustainable use and conservation of the resources by environmentalists, policymakers and lake managers.

1.3 Research questions

- ✓ What is the current state of physicochemical parameters in Lake Tinishu Abaya?

- ✓ How are the species composition, abundance and biomass of phytoplankton in Lake Tinshu Abaya?
- ✓ How are the zooplankton's species composition, abundance and biovolume in Lake Tinishu Abaya?
- ✓ How do the trophic interactions and energy flows among the major functional groups of the ecosystem function in Lake Tinshu Abaya?

1.4 Objectives of the study

1.4.1 General objective

The major goal of this research is to investigate water quality, phytoplankton and zooplankton composition, abundance and biomass, and to have a comprehensive description of trophic interactions and energy flow in the biotic components in Lake Tinshu Abaya.

1.4.2 Specific objective

The specific objectives are:

- ✓ To assess the physicochemical water quality parameters of Lake Tinshu Abaya.
- ✓ To determine phytoplankton species composition, abundance and biomass in the study lake.
- ✓ To determine zooplankton species composition, abundance, and biomass as biovolume in Lake Tinshu Abaya.
- ✓ To describe the trophic interactions and energy flow of LTA using Eco path with Ecosim model.

1.5 Significance of the study

There is lack of documented research on trophic status, energy flow and unified framework for ecological information of Lake Tinshu Abaya, which is essential for guiding management decisions. This study focuses on assessing the current status of water quality, phytoplankton and zooplankton composition and abundance, as well as trophic interaction and energy flow of Lake

Tinshu Abaya. It is crucial to examine these vital biological parameters in the lake ecosystem's trophic interaction and energy flow. Ultimately, this research will contribute to the expanding the breadth and scope of management objectives and strategies that can be explored for future management purposes

2. Literature review

Ecological modeling refers to the construction and use of mathematical or computational models to represent and simulate ecological systems. These models are designed to mimic the dynamics, interactions, and processes that occur within ecosystems(Jackson *et al.*, 2000).Ecosystem models have been rapidly growing topics in aquatic sciences since the 1990s, due to advancements in computer technology, increased needs for quantitative techniques, and management of aquatic ecosystems (Hauhs, 2013).

Ecosystem-based management (EBM) is a form of natural resource management that has grown consistently over the last two decades. It has emerged from the widespread feeling that traditional (single species) types of natural resource management have failed and that a new, more holistic way of understanding how ecosystems work is needed (Curtin and Prellezo, 2010).

It is a strategy for the integrated management of land, water, and living resources that promotes conservation and sustainable use in an equitable way. Thus, the application of the ecosystem approach will help to reach a balance of the three objectives of the Convention: conservation; sustainable use; and the fair and equitable sharing of the benefits arising out of the utilization of genetic resources. The ecosystem approach promotes conservation and equitable, sustainable management of land, water, and living resources. It relies on a scientific understanding of ecosystem structure, processes, functions, and interactions (Frid *et al.*, 2006).

Ecosystem-based management (EBM), an approach that acknowledge biodiversity, and consider ecosystem connections, has gained international popularity in recent years and several authors have used it for ecosystem management purpose(Long *et al.*, 2015). Correspondingly, EBM is a policy in Australia, Europe, the USA, and other countries (Heymans *et al.*, 2016).

In developing countries, the applicability of the ecosystem-based management approach is seldom because of difficulties to get the required information to run the approach. Implementing EBM is very costly and time and human resources, the cost of software, and the ecosystem services framework may not resonate with stakeholders already committed to biodiversity conservation (Wasson *et al.*, 2015).

Understanding the functioning of a complex ecosystem, and the possible impacts of different ecological changes on the system as a whole, calls for quantification of the trophic relationships between different groups in the system. This is one of the reasons why Polovina (1984) developed Ecopath modified by (Christensen and Pauly, 1992; Walters *et al.*, 1999; Christensen *et al.*, 2005), a steady-state model of trophic interactions in ecosystems. The relative simplicity of this model is apparent in its application to several marine, continental and freshwater ecosystems (Christensen and Pauly, 1993) .

2.1 Physico-chemical parameters

Water must be tested with different physicochemical parameters. Water does contain different types of floating, dissolved, suspended and microbiological as well as bacteriological impurities. (Gorde and Jadhav, 2013).Some physical test should be performed for testing of its physical appearance such as temperature, pH, turbidity, TDS, etc., while chemical tests should be perform for its BOD, COD, dissolved oxygen, alkalinity, hardness and other characters(Gorde and Jadhav, 2013).

2.1.1 Temperature

The metabolism of the aquatic ecosystem is controlled by water temperature. Summer head can cause fish deaths in water bodies because high temperatures lower the amount of oxygen available in the water, stressing aquatic ecosystems by reducing the capacity of water to contain necessary dissolved gases like oxygen. (Kumar and Puri, 2012).

2.1.2 Dissolved Oxygen

The amount of gaseous oxygen (O_2) dissolved in an aqueous solution is measured by analysis of the dissolved oxygen (DO). Through aeration (rapid movement), diffusion from the surrounding air, and as a byproduct of photosynthesis, oxygen enters the water. One of the most crucial factors in assessing the quality of water is the amount of dissolved oxygen present(Ibanez *et al.*, 2008).

The environmental impact of total dissolved solids gas concentration in water should not exceed 110% (above 13-14 mg/l). Concentration above this level can be harmful to aquatic life. Fish in waters containing excessive dissolved gases may suffer from "gas bubble disease". Adequate dissolved oxygen is necessary for good water quality(Kumar and Puri, 2012).

2.1.3 pH

Acidity of a solution is determined by pH(Cookson, 1974). The pH of a lake affects almost all chemical reactions. The buffering system of carbonate ($CO3\ 2-$) and bicarbonate (HCO_3 -) present in a lake and the amount of carbon dioxide in the water are two major factors influencing a lake's pH(Horne and Goldman, 1994).

2.1.4 Turbidity

Turbidity is formed by particles suspended or dissolved in water that scatter light, giving the water a cloudy or murky appearance. Sediment, particularly clay and silt, fine organic and inorganic materials, soluble colored organic compounds, algae, and other tiny creatures are examples of particulate matter(Kirk, 1985).Turbidity affects the physical, chemical, and biological condition of lakes by varying water temperature. The major sources of inorganic turbidity are runoff, shoreline erosion, and resuspension of bottom sediments(Howick, 1985).

Depending on the season, a lake may have detrimental impacts from high turbidity. Later in the summer, floating algae in lakes can obstruct other plants' ability to thrive by absorbing light. Fish that eat these plants and other animals may suffer as a result of this(Scheffer *et al.*, 2003).

Fish can also be impacted by high turbidity caused by algae because when a lot of algae die, oxygen is taken up during their decomposition, leaving less oxygen for the fish. Clay or large volumes of suspended dirt can block fish gills and cause immediate death.(Willemsen, 1980). Additionally, high turbidity may bury and destroy eggs placed on the bottom of lakes and rivers as well as make it difficult for fish to detect and catch food. Turbidity-causing particles may also be connected with pollutants and dangerous germs(Piper, 1982).

2.2 Chemical feature

2.2.1 Ammonia Nitrate and Nitrite

Nitrate ($NO3^{-)}$, the most highly oxidized and usually most abundant form of inorganic nitrogen, is the secondary nitrogen source for phytoplankton. Nitrite ($NO2$ -) is an intermediary and usually has insignificant concentrations (Horne and Goldman, 1994). Tiny quantities of $NH4$ $^{+}$ continue to exist in water because aquatic organisms excrete it, which algae quickly absorb. At these low concentrations, which are typically below the detection threshold, several plants may metabolize $NH4^+$. Recycling happens quickly. The $NH4^+$ exhaled by aquatic animals, according to can increase phytoplankton metabolism, however this $NH4^+$ source is rather little in comparison to that generated by bacterial breakdown(Horne and Goldman, 1994).

2.2.2 Phosphorus

According to Nürnberg and Peters (1984) a water system will experience an increased P load as a result of phosphorus (P) release from anoxic sediments or internal P load. As a result, there may be a shift from P to N limitation, which promotes eutrophication. The demand for oxygen in water and sediment rises as a result of increased production when more debris settles. In light of this, it is also likely that the area of sediment covered by anoxic water will grow. There is a chance that area P release rates will rise as a result of increasing sediment P concentration brought on by increased sedimentation. Internal stresses are significantly larger when these two increases are combined. Water systems are fertilized by internal P load, a self-improving process. Only winter and spring mixing returns phosphorus to the epilimnion in deeper water, whereas PO4 can be recycled from the sediments in shallow lakes or in the littoral zone(Horne and Goldman, 1994).

A wide range of pollutants are found in natural waterways as a result of erosive, corrosive, and weathering processes. Agricultural runoff is one of the nonpoint sources of pollution that has an impact on water quality. Poor animal husbandry techniques, overgrazed grasslands, excessive use of pesticides, plowing over irrigated fields, and fertilizer application are some agricultural practices that can affect the lake water quality(Khatri and Tyagi, 2015).

2.3 Phytoplankton

Phytoplankton is the first aquatic food chain to emerge. Phytoplankton are the primary producers and good indicators of the trophic status of aquatic ecosystems(Tadesse Fetahi *et al.*, 2014). They convert solar energy into chemical energy and release oxygen to the water body and the surrounding terrestrial environment through photosynthesis(Barupal and Gehlot, 2015). The productivity of an aquatic environment is directly correlated, with the density of phytoplankton(Benarjee and Narasimha, 2013) as they play an important role as primary producers and thus can affect higher trophic levels by providing nutritional bases for zooplankton and subsequently to other invertebrates, shell fish and finfish (Emmanuel and Onyema, 2007).

2.3.1 Chlorophyll a

The primary pigment utilized by phytoplankton to absorb light energy and transform it into biomass is called chlorophyll a (Chl a). Chlorophyll a measurements can be used to calculate the phytoplankton standing crop(Moss, 1969).

Chl a serves as a useful biomass estimate for phytoplankton since it is specific to plants and is simple to measure(Jakobsen and Markager, 2016). Phaeophytin a, another algal pigment, is quickly formed as chlorophyll a deteriorates. The quantity of phaeophytin a can be used to estimate how much of the lake's chlorophyll has been destroyed (Tett *et al.*, 1975). The quantity of phaeophytin a can be used to estimate how much of the lake's chlorophyll has been destroyed. Phaeophytin a is created when light and heat are applied to chlorophyll a. A longer processing period for the sample might result in higher phaeophytin a levels and lower chlorophyll a level. Additionally, it must be remembered that the chlorophyll a measurements only provide information on the phytoplankton crop that is now present and not the rate of primary production in the lake(Spalinger and Bouwens, 2003).

2.4 Zooplankton

The zooplankton of lakes is a key component of the ecology of water bodies because these organisms feed on phytoplankton, recycle the nutrients through excretion, and represent an important prey to many predators(Echaniz *et al.*, 2012). In freshwater environments, zooplankton plays a relevant role in energy transfer and nutrient transport and regeneration, owing to their position in the food web as the main direct consumers of phytoplankton(Armengol and Miracle, 1999). Zooplankton communities play an important role in the food web structure and in ecosystem health by feeding on phytoplankton, and they serve as food for higher trophic fishes and many carnivorous invertebrates. Their direct and rapid response to many physical, chemical, and biological changes in aquatic ecosystems means that zooplankton represents a useful indicator that could reflect the changes in aquatic ecosystems caused by pollution and various other disturbances (Marques *et al.*, 2007).

2.5 Benthic Macro Invertebrates

One of the most important components of lake ecosystems are benthic macro invertebrates. (Venkatesharaju *et al.*, 2010). In aquatic food webs, they serve as an important link between primary producers, detrital deposits, and higher trophic levels. (Stoffels *et al.*, 2005). Eutrophication may be indicated by benthic invertebrates, but there are several other modes of lake degradation. (Hooda *et al.*, 2000). As a result, classifications based on benthic invertebrates are expected to differ from those based on planktonic communities, particularly in lakes subjected to multiple impacts. (Hooda *et al.*, 2000).

Water quality and the overall health of aquatic ecosystems can be assessed using macro invertebrate measurements (Tampo *et al.*, 2021). Benthic macro invertebrates are the most widely utilized bio-indicators of water quality among the biological populations(Bonada *et al.*, 2006), because several characteristics make them easy to study and show clear responses when faced with adverse environmental conditions(Moreno *et al.*, 2009). There are a number of biotic scores based on benthic macro invertebrates have been developed and are successfully used for the bio assessment of rivers in different parts of the world such as (Birk *et al.*, 2012; Elias Dadebo *et al.*, 2014b; Aschalew Lakew and Moog, 2015)

2.6 Macrophytes

Aquatic macrophytes play an important role in structuring communities in aquatic environments. These plants provide physical structure, increase habitat complexity and heterogeneity and affect various organisms like invertebrates, fishes and waterbirds. The complexity provided by macrophytes has been exhaustively studied in aquatic environments(Thomaz and Cunha, 2010).

Macrophytes *colonize* many different types of aquatic ecosystems, such as lakes, reservoirs, wetlands, streams, rivers, marine environments and even rapids and falls (e.g., family Podostomaceae). This variety of colonized environments results from a set of adaptive strategies achieved over evolutionary time. Primary production of macrophytes can surpass that of other aquatic primary producer(Wetzel, 2001; Kalff, 2002).

Macrophytes generally colonize shallow ecosystems where they become important components, influencing ecological processes (e.g., nutrient cycling) and attributes of other aquatic attached assemblages (e.g., species diversity)(Thomaz and Cunha, 2010).

They are primary producers and provide habitats and refuges for periphyton, zooplankton, other invertebrate species, fish and frogs. They also play key functions in biogeochemical cycles (e.g. organic carbon production and phosphorous mobilization) directly influencing hydrology and sediment dynamic of freshwater ecosystems through their effects on water flow. Macrophytes contribute to maintain key functions and related biodiversity in freshwater ecosystems, but also through the services they provide directly to man(Bornette and Puijalon, 2009).

Aquatic macrophytes use nutrients such as nitrate and phosphate, limiting the amount of nutrients available for phytoplankton uptake(Dennison *et al.*, 1993). Additionally, they can retain the sediment, which further sequesters nutrients and reduces turbidity (Dennison *et al.*, 1993; Horppila and Nurminen, 2003). As macrophytes effectively compete with phytoplankton for nutrients in the water column, algal blooms and the associated turbidity are limited, preventing decreases in water quality (Bakker *et al.*, 2010).

2.7 Fish

Similar to other organisms, fish requires energy to grow, survive and reproduce(Lagler *et al.*, 1977) .In a freshwater habitat, energy could be obtained from a variety of food such as detritus, plants and animals. Fishes in tropical streams have diverse feeding behavior compared to temperate fishes. Knowledge on the feeding habit and food preference of freshwater fish is important for maintenance and conservation of the species either in captivity or in its natural habitat(Braga *et al.*, 2012).

The study of the food and feeding habits of freshwater fish species is a subject of continuous research since it constitutes the basis for developing successful fisheries management program on fish capture and culture(Olurin and Aderibigbe, 2006).

The Feeding behavior of fish is the major important factor affecting their nutrition and growth (Afrah and Maktoof, 2013). Food availability determined the well-being of fishes, and their reproductive potentialities in any aquatic ecological system and the weight and size of fish reflect food availability in the aquatic ecosystem (Elias Dadebo *et al.*, 2014a).

Feeding is the dominant activity of the entire life cycle of fish (Royce, 1972). Therefore, the study of food and feeding habits of a fish is very important. This is also essential for any fishery management. Food and feeding habit of fish are important biological factors for selecting a group of fish for culture in ponds to avoid competition for food among themselves and live in association and to utilize all the available food (Khaing and Khaing, 2019).

3. Methodology

3.1 Study area

Lake Tinshu Abaya (also called Lake Silte Abaya) is located at 7 0 29'03.65" N latitude and 380 03'17.79" E longitude at an altitude of 1835 meters a.s.l. The lake is relatively small in size, with a total surface area of 12.53 km^2(Kassahun Asaminew *et al.*, 2011). It is a shallow lake with maximum and mean depths of 1.3 m and1.0 m, respectively. The rainfall of the lake's region ranges from 500 to1300mm.The lake's eastern shore is bounded by a mountain. It has two major feeder rivers: The Dacha River (northern corner) and Bobodo River (southern corner). The northern side of the Badober River acts as the lake's spillway. Badober River (northern side) acts as an outlet for the lake.

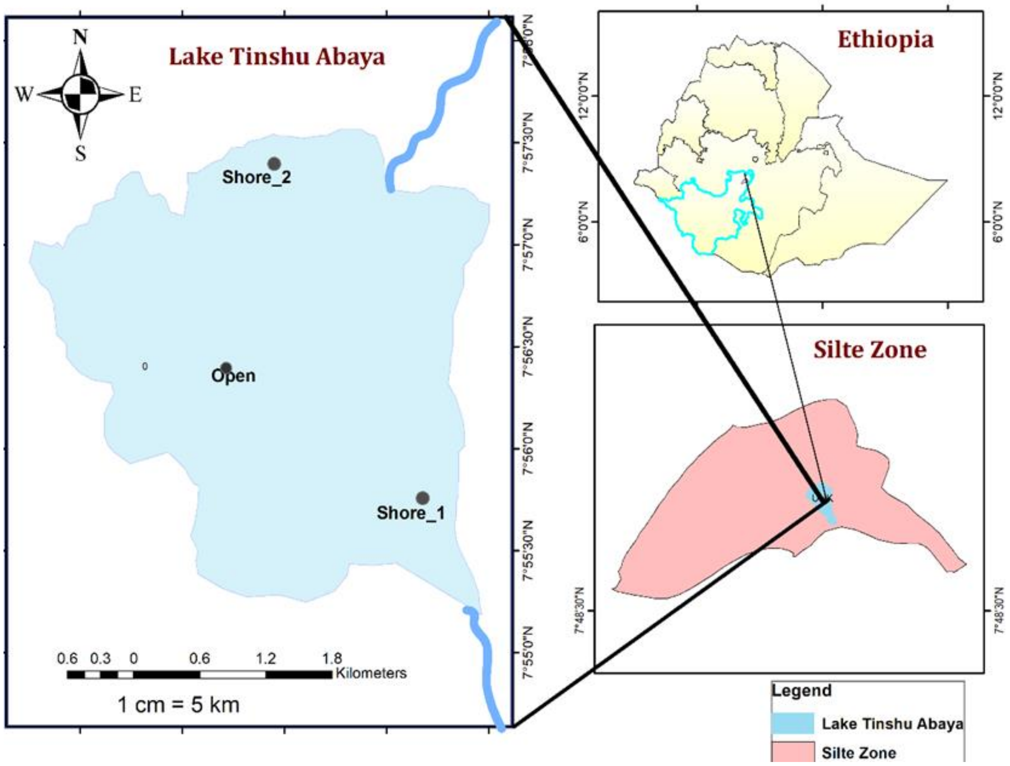

Figure 3.1: Map showing the location of Lake Tinshu Abaya

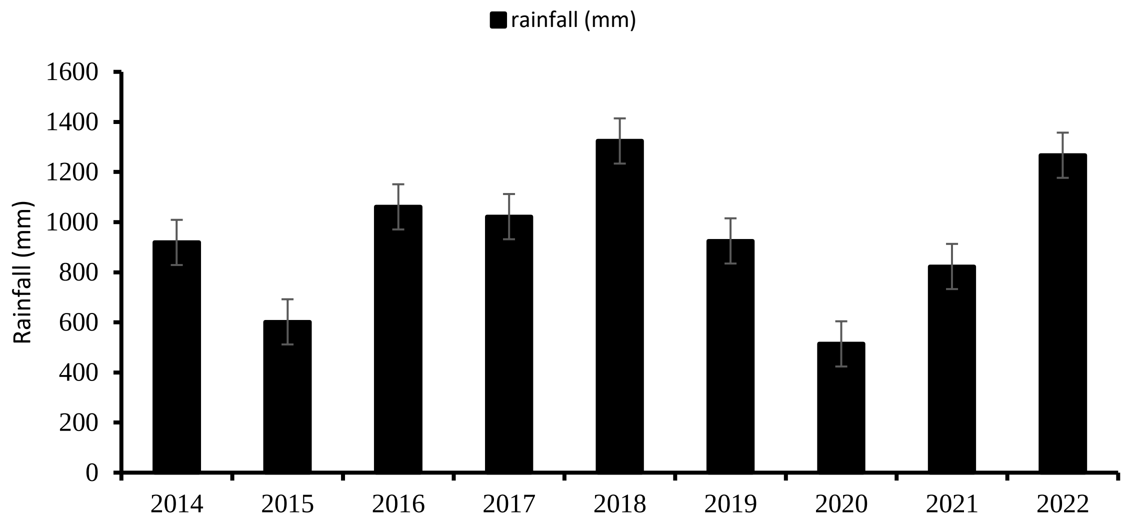

Figure 3.2: The annual rainfall(mm) pattern of the study area during the last 9 years.

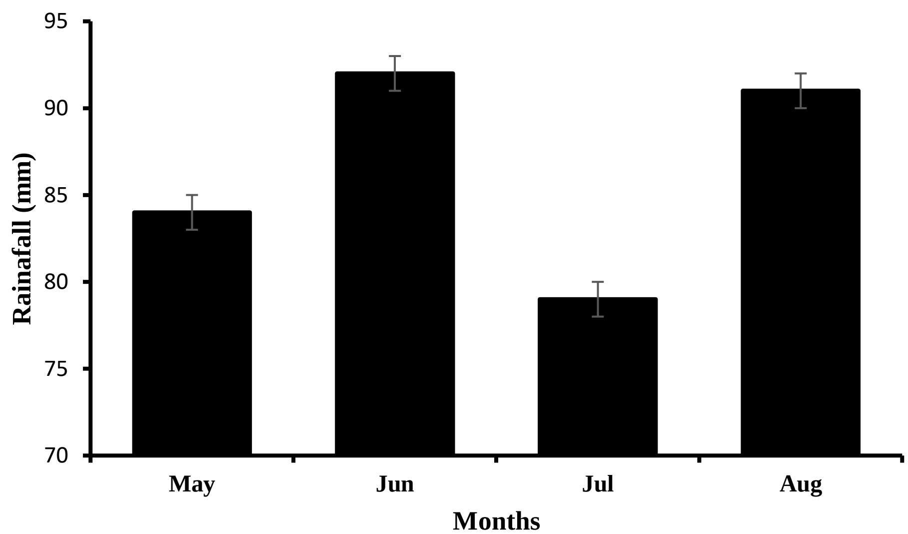

Figure 3.3: Mean monthly rainfall for the sampling months at 2022.

3.2 Sampling protocol

The study was conducted from May 2022 G.C to August 2022 G.C. Three sampling sites were carefully selected to ensure good representation of microhabitats and limnetic and littoral zones. Samples were collected for three consecutive months using plastic containers, stored in an ice box and immediately transported to the Seyoum Mengistoue Limnology Laboratory of Addis Ababa University for analysis.

3.3 Sample size and sampling techniques

The water sampling sites were chosen using both non-probability (on purpose) sampling approaches to include areas with significant pollution and regions near possible pollution sources, and probability (simple random selection) sampling techniques to provide a representative sample. and a total of nine samples was collected for the determination of physical, chemical, and biological water quality parameters of Lake Tinshu Abaya.

Table 3.1: Sampling sites and their description

Sites	**GPS**	**Description**
Open_1	7.94281ºN, 38.367º E	Open site. The site is open with a max depth of about 1.3m.
Shore_1	7.932464º N 38.37 º E’	The sedimentation from the mountain. The site is offshore with a depth of about 0.8 m
Shore_2	7.95603º N 38.36429º E	Pollution from agricultural activities, and anthropogenic activity. The site is offshore with a depth of about 0.8 m.

3.4 Preliminary data collection

Prior to collecting the samples, a survey of the study area was conducted using previously published papers to gather information on water quality influences, the abundance of phytoplankton and zooplankton, as well as the composition and biomass status of Lake Tinshu Abaya. Both primary and secondary data were compiled and prepared as input for Ecopath**.**

3.5 Physicochemical parameters

Dissolved oxygen (DO), pH, temperature, conductivity, and turbidity, were measured *in situ* using a HACH multi meter probe (HQ40d) three times per sampling to make certain and Secchi depth (Z_{SD}) was measured by a Secchi disk of 20 cm diameter.

Figure 3.4: Measuring the physico-chemical parameters of the LTA at a sampling site

3.5.1 Nutrient analysis in the laboratory

Composite samples were prepared by mixing, in equal proportions, from subsurface areas. The water samples were filtered through Whatman Glass Fiber Filter papers (GF/F), and the filtrates were used for the colorimetric determination of dissolved inorganic nutrients according to the standard methods described in APHA (1999). Ammonia-Nitrogen (NH_3^+ NH_4^+-N) was determined by the Phenate method, Nitrite-Nitrogen (NO_2^--N) by diazotization with sulphanilamide and coupling with N-1-naphthyl ethylenediamine di-HCl, Nitrate-Nitrogen (NO_3^--N) by the Salicylate method, Soluble Reactive Phosphate-Phosphorous (PO_4^{3-}-P) by the Ascorbic acid method and Dissolved molybdate reactive Silica (SiO_2) by the Molybdosilicate method. However, unfiltered water samples were used for the estimation of Total phosphorus (TP), after per-sulfate digestion, with the Ascorbic acid method. The absorbance of the colored solutions was read using a Jenway 6405 spectrophotometer at appropriate wavelengths (nm) in the Limnology laboratory.

Total Suspended Solids (TSS) was determined gravimetrically as the dry weight of seston filtered onto a glass fiber filter paper (GF/F, 47mm pore size), which was pre-dried at 105°C and subsequently dried with seston for 1 hour at the same temperature. The following formula was used to calculate TSS. (Estefan *et al.*, 2013).

$$\text{TSS (mg/L)} = \frac{(W2\_W1)\ X\ 1000}{V}$$

Where: W_1 = Weight of dried clean filter paper

W_2 = Weight of dried clean filter paper and seston

V = Volume of water sample used for filtration

For Tropic index (Carlson, 1977) were used

Carlson's Trophic State Index (TSI): TSI = 60 + 6.25(log10(chlorophyll-a)) - 31.2(log10(Secchi depth)) + 10.4(log10(total phosphorus)) - 0.44(log10(total nitrogen))

3.6 Determination of phytoplankton species composition, abundance and biomass

3.6.1. Identification and enumeration of phytoplankton

Composite samples were prepared from samples collected from subsurface area and placed in plastic sample containers. The collected samples were preserved with Lugol's iodine solution, and used for the identification of taxa using various taxonomic guides(Edward and David, 2010; Komárek, 2013; Van Vuuren *et al.*, 2006; Verlencar and Desai, 2004). Enumeration of phytoplankton was performed using a Sedgewick-Rafter cell 30 square per sub sample were counted and an inverted microscope, as described in (Hotzel and Croome, 1999).

3.6.2 Determination of phytoplankton Abundance

According to (Eyo *et al.*, 2013) and (Rahman *et al.*, 2018) abundance was determined using the formulae below;

- Abundance (N) = (A* 1000* C)/ (V*F*L)

Where N = Number of phytoplankton in cells per litter

A = Total number of phytoplankton cells counted

C = Volume of final concentrate of samples in ml,

V = Volume of a field in ml

F = Number of the fields counted   L = Volume of original water in L

3.6.3 Phytoplankton biomass estimation

The concentration of Chlorophyll-a (Chl-a) was used to estimate phytoplankton biomass. 250-500 mL aliquots of water were filtered using glass fiber filter paper (Whatman GF/F). A scissor is used to break down the filter paper, releasing phytoplankton cells and chlorophyll-a. The filter paper containing phytoplankton cells is put in an appropriate container in to the solvent acetone. A homogenizer probe was used to homogenize the mix of solution. After extraction of the pigment in cold 90% acetone and clearing by centrifuging for 10 minutes at 3000 rpm, the absorbance of chlorophyll-a was measured spectrophotometrically. Absorbance of pigment extracts was measured at 665 and 750 nm using 90% acetone as a blank.

The concentration of *Chl-a* was calculated according to (Lorenzen, 1967). No correction was made for degradation products. *Chl-a* values was converted to wet weight, as this is required by EwE; subsequent conversion of *Chl-a* to carbon and then carbon to wet weight was, therefore, made. The carbon to chlorophyll ratio of 40:1 was used as the factor for converting Chl-a to carbon and concentration of carbon was equated to 10 % of wet weight (Jones, 1979) .

3.7 Identification of zooplankton species

For the identification of zooplankton taxa, samples were collected by means of horizontal net hauls with tow nets of 70 µm from subsurface and preserved in a 4% formaldehyde solution. The volume of water filtered was determined using the following formula:

$$V = \pi r^2 d$$

Where V is the volume of the water filtered (m^3)

r is the radius of the net (m).

d the distance through which the plankton net was towed (m).

3.7.1 Estimation of Abundance and biovolume of Zooplankton

In order to estimate the zooplankton's abundance and biovolume, samples were poured into a cylinder and allowed to concentrate for a short period of time. Sub- samples from the concentrated samples were added to a petri dish with grid lines and counted using a wild dissecting microscope at a 40X magnification.

Species were identified using various identification keys (Voigt and Koste, 1978; Van de Velde, 1984; Danielle Defaye, 1988; Dussart and Fernando, 1988; Fernando, 2002).Adult Cladocerans Copepods, Rotifers, and immature Copepods (Nauplius and copepodites) were counted as separate groups.

The final estimates of the populations (individual's m^3 of lake water) were computed using the formula of (Edmondson and Vinberg, 1971).

The lengths, widths, and heights of 10 individuals of each species per sample were measured for the estimation of biovolume(Binggeli *et al.*, 2011; Rocha *et al.*, 2021).

3.8 Ecopath model description

Using Ecopath with Ecosim (EwE) software version 6.6.7, a mass balance trophic model of LTA was constructed to quantify trophic interactions and energy flows among the compartments of the ecosystem. This approach is based on two master equations, one describing the production term and the other one describing the consumption term. The production term in Ecopath is described as: Production is equal to the sum of fisheries catches, predation mortality, biomass accumulation, net migration and other mortalities.

Christensen *et al.* (2005) put this in a more formal way as:

$$\mathrm{B_i}\left(\frac{\mathrm{P}}{\mathrm{B}}\right)_{\mathrm{i}}\mathrm{EE} = \sum_{\mathrm{j=1}}^{\mathrm{n}} \mathrm{B_j}\left(\frac{\mathrm{Q}}{\mathrm{B}}\right)_{\mathrm{j}}\mathrm{DC_{ij}} + \mathrm{B_i}\left(\frac{\mathrm{P}}{\mathrm{B}}\right)_{\mathrm{i}}(1-\mathrm{EE_i}) + \mathrm{EX_i}$$

where $B_i$is the biomass of group *i* (in t km^{-2} fresh weight); (*P/B)i* is the annual production/biomass ratio of *i* equal to the total mortality coefficient (*Z*) in steady-state conditions (Allen, 1971); *EEi* is the Eco trophic efficiency representing the part of the total production consumed by predators or captured in the fishery or exported; *Bj* is the biomass of the predator group *j; Q*/*Bj* is the annual food consumption per unit biomass of the predatory group *j; DCji*, is the proportion of the group *I* in the diet of its predator group *j*; *EXi***,** is the export or catch in fishery of group *i*, that is assumed to be exploited in the fishery (Christensen *et al.*, 2005).

Consequently, for each entity (functional group) modeled in Ecopath, information is required on production/biomass (*P/B*), consumption/biomass (*Q/B*), biomass, proportion of habitat area occupied, biomass in habitat area ($t.km^{-2}$), diet composition and fishing mortality.

Similarly, the energy input and output of all living groups must be balanced and for this, (Christensen and Pauly, 1992; Christensen *et al.*, 2005); described the second master equation of Ecopath as (2)

$$Qi=Pi+ Ri +Ui$$

Where Q is consumption, P – production, R – respiration and U – unassimilated food of the ith group. The two master equations together ensure the energy balance of each group.

3.8.1 Parameterization

The model represents an annual average situation of the LTA ecosystem. Biomass value was obtained from the lake survey and information available in the literature. Production/biomass ratios (*P*/*B*), consumption/biomass ratios (*Q*/*B*), production/consumption ratios (*P*/*Q*), and unassimilated/ consumption ratios (*U*/*Q*) was taken from the literature or obtained from the application of empirical equations using length and weight data (Palomares and Pauly, 1998; Christensen and Walters, 2004).

3.8.2 Input parameters

In Eco path, functional groups can be composed of species, groups or ontogenetic fractions of a species (Christensen and Walters, 2004). Similarity of the habitat, diet and life history characteristics was considered.

Table 3.2 Input parameters of Lake Tinshu Abaya

Group/Parameters	**Methods of Estimation/Determination**
Phytoplankton	

Biomass	Average phytoplankton biomass was determined from monthly sample collections
Production/ biomass	Productivity values were estimated using the oxygen method and also data from previous studies on the lake were used.
Zooplankton	
Biomass	Biomass was estimated from monthly sample collection
Production /biomass	After identifying potential groups using the Egg ratio method and the production method, their growth rates were estimated in the laboratory and data were obtained from the previous studies made on the lake
Consumption /biomass	Q/B for zooplankton was estimated based on assumed gross food conversion efficiency (P/Q) of 0.2 (Christensen *et al.*, 2005).
Detritus biomass (D)	
Biomass	Detrital biomass was calculated as a function of primary production and euphotic depth by employing the following relationship suggested by Christensen and Pauly (1993). LogD = 0.954 log PP + 0.863 log E − 2.41 where D= detrital biomass (gC/m2); PP = primary production (in gCm−2 year−1);
	E = euphotic depth in meter (0.082m).
Zoobenthos	

Biomass	Weed bed samples were taken with a strong metal hand net and the average zoo benthos biomass was calculated
Production /biomass	Was estimated using the empirical formula of (Brey 1999). Log P/B = 1.672 + 0.993 log (1/Amax) − 0.035 log (Mmax) −300.447 1/ (T + 273) Where A_{max}: maximum age (per year); M_{max}: maximum individual body mass (g DM); T: bottom water temperature (^{0}C).
Consumption /biomass	Was estimated based on assumed gross food conversion efficiency (P/Q) of 0.2 (Christensen *et al.*, 2005).

3.8.3 Fisheries yield (Y)

Catch statistics for the fishery target fish groups was obtained from the fisheries records through the request made by the Silte Woreda Bureau of Agriculture.

Table 3.3: Input parameters of the LTA Fishery

Parameters	Methods of estimation
Biomass	The biomass (B) was estimated assuming equilibrium conditions, such that: B = Y/F where Y is yield in $t.km^{-2}.\ yr^{-1}$ and F is the coefficient of fishing mortality. F is the difference between total and natural mortalities: F = Z – M, assuming that Z is equal to P/B as indicated by (Allen, 1971). Natural mortality, M, was computed using the predictive formula of Pauly (1980)
	Ln (M) = −0.0152 − 0.279 ln (L∞) + 0.6543 ln (K) + 0.463 ln (T) where L∞ and K are parameters of (Von Bertalanffy growth function VBGF); and T = annual mean surface water temperature of the lake.
Production/ Biomass	The P/B ratio of the fish groups was estimated using length-frequency distributions. The software was used to estimate the growth parameters of the Von Bertalanffy (1938). Growth Function, *i.e.*, the asymptotic length, L∞, and

	the growth coefficient, K, which are needed for P/B computation (Gayanilo *et al*., 2002).
Consumption /Biomass	Data was obtained from Fish Base (Froese and Pauly, 2007) and studies made by various investigators. The food consumption per unit of biomass (Q/B was estimated using the multiple regression formula of (Palomares and Pauly, 1998) **Log (Q/B) = 5.847 + 0.280Log (P/B) – 0.152LogW– 1.360T' + 0.062A + 0.510h + 0.390d** where: W∞ = fresh weight; T' = 1000/Kelvin (Kelvin = T° C + 273); A = the aspect ratio of the caudal fin, indicative of metabolic activity and expressed as the ratio of the square of the height of the caudal fin and its surface area, A, was obtained mainly from Fish Base (www.fishbase.org) (Froese and Pauly, 2007).
	The parameters h and d refer to diet, h = 1, d = 0 for herbivorous; h = 0, d = 1 for detritivores; h = 0, d = 0 for carnivorous.
Yield	The evaluations of actual catches are based on landing statistics for each species or group of species. The total annual catch from the ecosystem was estimated from the Data collected by Silte Agricultural Fisheries Center.

3.8.4. Ecological grouping

Previous studies on the lake were also evaluated in order to categorize organisms into functional groups. In Ecopath, functional groups can be made up of species, groups, or ontogenetic divisions of a species (Christensen and Walters, 2004).

Phytoplankton from the current study and (Yirga Enawgaw and Brook Lemma, 2018a),; Zooplankton from current study and Yirga Enawgaw and Brook Lemma (2018); Zoo benthos and

Tilapia(Yirga Enawgaw and Brook Lemma, 2019) and (Mathewos Hailu *et al.*, 2022) and Birds (Moreau *et al.*, 1993).

Table 3.4: Possible ecological grouping

No	Functional group	Group members
1	Waterfowls (Birds)	Pelicans, Cormorants, Kingfishers
2	Tilapia	*Oreochromis niloticus*
4	Macro-zoo benthos	Gastropods, Odonata, Diptera, Hirudinae (Leeches), and Oligochaetes
5	Carnivores zooplankton	*Mesocyclops aequatorialis*
6	Herbivores zooplankton	*Thermocyclops decipiens*, Cladocerans and rotifers
7	Phytoplankton	Cyanophyta, Chlorophyta, Bacillariophyta and Euglenophyta

3.9 Balancing the model

Eco trophic efficiency (EE) and gross efficiency (GE) were used as a guide for balancing the model, whereby EE values should not be larger than 1 and GE values should not exceed 0.3 (Christensen and Pauly, 1992).

3.10. Trophic interactions

Once the model is balanced, a flow diagram was constructed. A flow diagram shows the trophic interactions and the flow of food energy from one functional group to the other.

Graphically, different boxes representing each functional group of the ecosystem are arranged in a hierarchy based on their computed trophic levels (y-axis) along the x axis in such a way that they don't overlap. The different groups were then connected with networks showing their trophic interactions and energy flows (Christensen *et al.*, 2005).

The concept of trophic level (TL) was introduced long ago by Lindeman (1942), who assigned integers or discrete values to the different producers and consumers of a system. In EwE, the TLs are not necessarily integers (1, 2, 3…) as proposed by Lindeman, but can be fractions (e.g., 1.2, 2.5, etc.). A standard technique is used in EwE to assign trophic levels (TL) based on definitions. A trophic level of 1 is ascribed to producers and detritus. whereas, consumers are allocated a trophic level of 1 plus the weighted average of their prey's trophic levels(Christensen *et al.*, 2005).

Another method divides the entire ecosystem into several trophic levels is also known as the "census Lindeman" approach, defines trophic levels as distinct levels within the system(Christensen *et al.*, 2005).This analysis then allowed for calculation of trophic transfer efficiencies (%) expressing how energy is transferred efficiently from one trophic level to the next. To assess the state of the fishery, mortality estimates were computed with the EwE routine and the exploitation rate was then calculated from computed fishing and total mortality rates.

3.11 The omnivore index

The 'omnivory index' was introduced in 1987 in the initial version of the Ecopath II software. This index (OI) is calculated as the variance of the trophic level of a consumer's prey groups. Thus

$$\mathrm{OIi}=\sum_{i=1}^{n} TLj-(TLi-1))^{2}.\mathrm{DCij}$$

where OIi is the omnivory index of predator i, TLj is the trophic level of prey j, TLi is the trophic level of the predator i, and DCij is the proportion of prey j constituting the diet of predator i. When the value of the omnivory index is zero, the consumer in question is specialized, i.e., it feeds on a single trophic level. A large value indicates that the consumer feeds on many trophic levels. The omnivory index is dimensionless. The square root of the omnivory index is the standard error of the trophic level, and a measure of the uncertainty about its precise value due to both omnivory and sampling variability.

3.12 Niche overlap index

The niche overlap index is a quantitative metric that evaluates how much resource use by various species or individuals within a population overlaps or is similar to one another. It gives insight into potential ecological interactions, including competition or predation, between organisms sharing similar resources. This index ranges from 0 to 1, where 0 indicates no overlap (completely separate resource utilization) and 1 indicates complete overlap (identical resource utilization)(Hurlbert, 1978; Loman, 1986).

The niche overlap index of two species/groups (j) and (k), Ojk, which has been used for many descriptions of niche overlap, can be estimated from

Ojk=$\sum_{i=1}^{n}(pji * pki)/\sqrt{}(\sum_{i=1}^{n} p^2jip^2ki)$

Where Pji and Pki are the proportions of the resource (i) used by species (j) and (k), respectively. The index is symmetrical and assumes values between 0 and 1.

3.13 Connectance index

The Connectance index (CI) is, for a given food web, the ratio of the number of actual links to the number of 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. The number of possible links in an Ecopath model can be estimated as $(N-1)^2$, where N is the number of living groups. It has been observed that the actual number of links in a food web is roughly proportional to the number of groups in the system.

Thus, CI $N/(N-1)^{\tilde{2}} \approx 1/(N-1)$

which defines a hyperbolic relationship. (Odum and Barrett, 1971) expected food chain structure to change from linear to web like as systems mature. Hence, the Connectance index can be expected to be correlated with maturity. The value of the Connectance index is, at least in aquatic systems, largely determined by the level of taxonomic detail used to represent prey groups, and this precludes meaningful intersystem comparisons. The System omnivory index is suggested as an alternative.

3.14 Key stone Species

Keystone species were initially defined by (Paine, 1966) as "a predator supporting high species diversity and complicated interspecific feeding connections in the ecosystem."

Key stoneness (KS) has been incorporated in the EwE network analysis module. The EWE software suites came with three indices. The initial index KS was suggested by (Libralato *et al.*, 2006)

KS1=log| εi, x (1pi))

The second index KS2 was proposed by(Power *et al.*, 1996).

KS_2 = log (εi, x(1/pi))

In both equations, εi is a measure of trophic impact and p is measure of biomass.

The third indicator was suggested by (Valls *et al.*, 2015). The third index (KS3) is a hybrid of KS1 and KS2. The equation is generally written as

KS3= log [IC* BC]

Here, IC stands for impact component and BC stands for biomass component.

3.15 Trophic aggregation

In ecological modeling, trophic aggregation is the process by which species in an ecosystem are classified or aggregated into trophic levels based on their feeding linkages. It simplifies the complex food web by categorizing organisms according on where they fit in the trophic hierarchy. For the mathematical calculation of fractional trophic levels, which aggregates the whole system into discrete trophic levels by cense Lindeman, which is based on a technique provided by Ulanowicz (1995).

3.16 Network Analysis

Network analysis, as proposed by Ulanowicz (1986), was used to analyze ecosystem properties based on four indices: total system throughput (*T*), ascendency (*A*), development capacity (*C*), and system overhead (ϕ). Ascendency is a measure of the average mutual information in a system, scaled by system throughput, and is used as an ecosystem maturity index whose maximum limit is the development capacity. The difference, *C*−*A*, is known as system overhead (ϕ) and represents the system strength in reserve to withstand unexpected perturbations.

The closer ascendency is to development capacity, the more the ecosystem is mature and diverse (Ulanowicz, 1986; Christensen et al., 2004). However, when overhead is closer to development capacity, the system is less mature, but more resilient (Ulanowicz, 1997). Total throughput was also estimated by group and the fractional trophic level was calculated, as proposed by Odum and Heald (1975). Ascendency network indices was used as a measure of ecosystem development and maturity (Odum, 1969) and other indices were computed.

3.17 Mixed trophic impact

The direct and indirect trophic impacts of a change in the biomass of a group that will have on the biomass of other groups in a system was graphically presented using the Mixed Tropic Impact (MTI) routine of EwE (Ulanowicz and Puccia, 1990; Christensen *et al.*, 2005). An increase in the biomass of a predator group would result in a negative impact on its prey groups, while positively impacting the prey of its preys through a reduction in predation pressure. Similarly, groups that have similar trophic niches would have an impact on each other through competition for feed resources. Mixed trophic impacts, was calculated with the Leontief matrix method developed for ecosystem analysis by Ulanowicz and Puccia (1990).

3.18 The pedigree index

A "pedigree" was implemented to be certain with a data-intensive process including defining a set of probability distributions for all input parameters, including the diet composition matrices. This implementation was made to increase the process's clarity. (Funtowicz and Ravetz, 1990).

The pedigree of an Ecopath input categorizes the origin of a given input (the type of data on which it is based), and specifies the likely uncertainty associated with the input, i.e. the reliability of the data.

Based on the options selected for each parameter for each group a pedigree index, P, was calculated the product of all the pedigree parameter specific indices. The P scales between 0 and 1 (inclusive). Therefore, a measure of fit will also be used,

$t^* = P \cdot \sqrt{(N-2)} / \sqrt{1-P}$ where N is the number of living groups in the given model. The pedigree index values are also used to calculate an overall pedigree index for a given model. The index values for input data scale from 0 for data that is not rooted in local data up to a value of 1 for data that are fully rooted in local data.

Based on the individual index value an overall 'pedigree index' P is calculated based on

$$P=\Sigma \frac{Iij}{n}$$

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 represent either B, P/B, Q/B, Y or the diet.

3.19 Statistical analysis

The significance of spatial and temporal variations in physicochemical, phytoplankton and zooplankton data were analyzed using one-way Analysis of Variance (ANOVA) at 95 % significance level ($P < 0.05$). Causal relationships (correlations) among physicochemical and biological parameters were assessed and descriptive statistical analysis was made each using R software version 64.1.3.

3.20 Possible output of the research

Information on the composition, abundance, and biomass of phytoplankton and zooplankton in LTA has been updated, while the current state of food web interactions, and energy flow was revealed and possible management measures were recommended.

4.Results

4.1 Physico-chemical parameters of the Lake

Physical and aggregate chemical parameters

The water temperature of LTA was highest in May at shore_1 (25.01±0 °C) followed by that recorded for Shore_2 (24.3±05°C). Nevertheless, the lowest temperature was recorded in August at the Open water site (20.7±0°C) followed by that observed at Shore_1 (20.83±0.04°C). The temperature of the study lake showed marked temporal and spatial fluctuations (ANOVA; p <0.05).

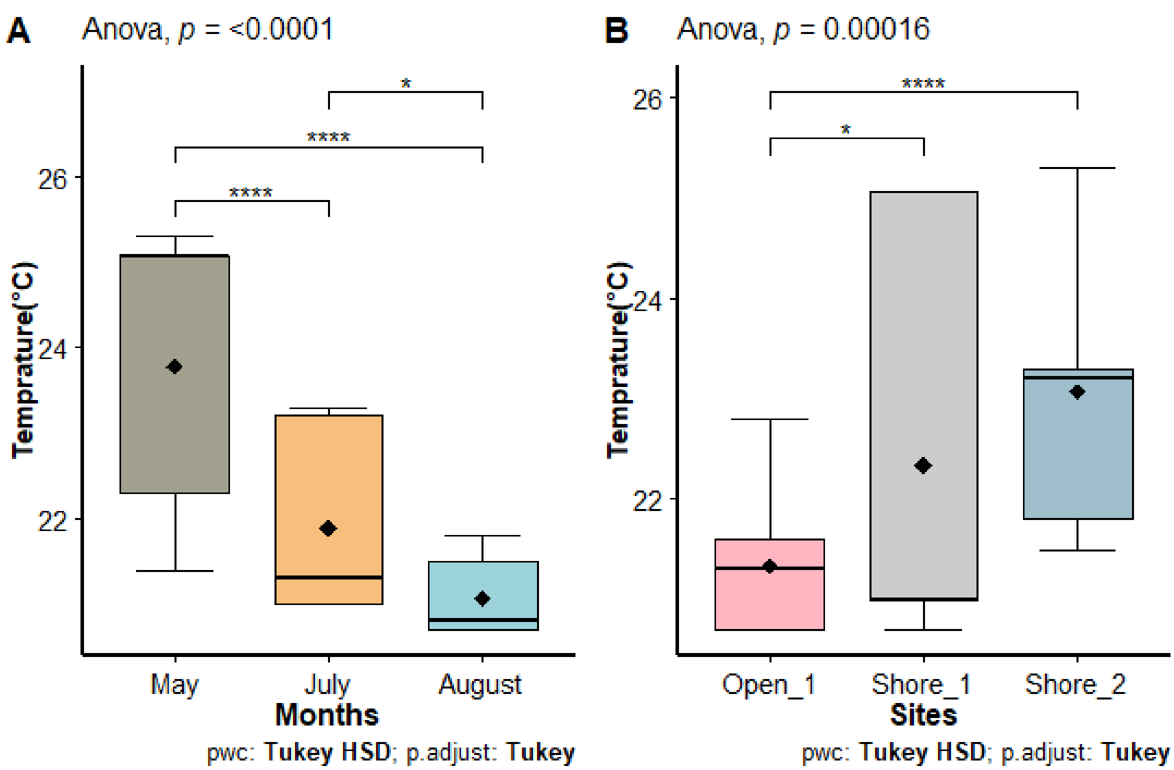

Figure 4.1: Box-and-whisker plot of Temperature measured at the three sampling sites during the three sampling months. (A) Temprature during Months (B) Temprature among Sites.

Dissolved Oxygen concentration was highest in May at Shore_2 (Mean =13.86 ± 0 mg/l) followed by at Shore_1 (Mean =10.37±0 mg/l). Nevertheless, the lowest dissolved oxygen concentration was observed in the July at Shore_1 (Mean =7.42±0.06 mg/l) followed by that recorded in August for Shore_2 (Mean =7.63±0.01 mg/l). Dissolved oxygen in the study lake showed marked temporal and spatial fluctuations (ANOVA; $P<0.05$).

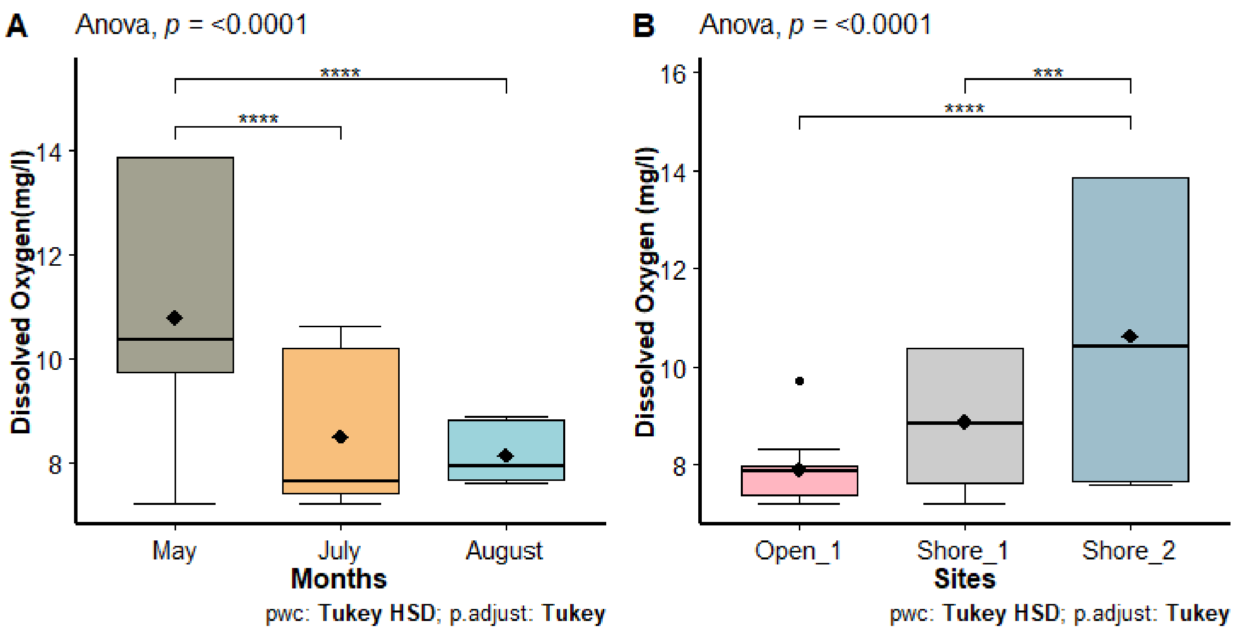

Figure 4.2: Box-and-whisker plot of Dissolved Oxygen concentration measured at the three sampling sites during the three sampling Months. (A) Dissolved Oxygen during Months and (B) Dissolved Oxygen among Sites.

The pH of the study lake was highest in July at Shore_2 (Mean =10.04±0) followed by Shore_1 (Mean =10.3±0.2). Nevertheless, the lowest level of pH was recorded in August at Shore_1 (Mean = 9.77±0.0) followed by at the Open water and Shore _2 sites (Mean =9.88±0.01). The pH of the study lake showed marked temporal fluctuations (ANOVA; $p<0.0001$) although there was no significant spatial variation (ANOVA, $p=0.66$).

The Total Alkalinity of the lake was highest in July at the Open water site (Mean =6.67±0.21 meq L^{-1}) and lowest in August at Shore_2 (Mean =5.47±0.21 meqL^{-1}).

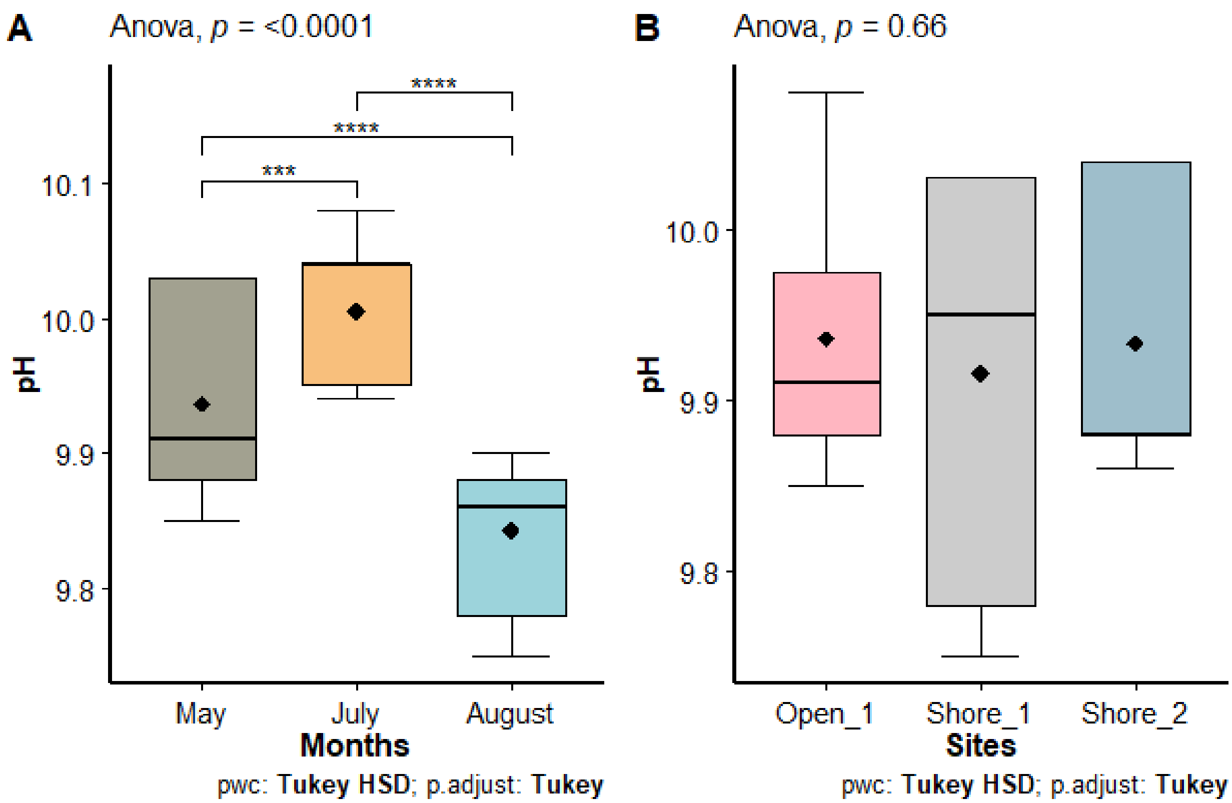

Figure 4.3: Box-and-whisker plot of pH recorded at the three sampling sites during the three sampling Months. (A) pH during Months (B) pH among Sites.

The Electrical Conductivity of the study lake was highest in May at Shore_2 (Mean =1505±0µS cm^{-1}) followed by that observed in July at Shore_2 (Mean =1502±0.29µScm-1). Nevertheless, the lowest Electrical Conductivity was recorded in August at the Open site (Mean =1246±1.04µScm-1) followed by at Shore_1 (Mean =1253±0.0µScm-1). The Electrical Conductivity of the study lake showed marked temporal fluctuations (ANOVA; P<0.001) although there was no significant spatial variation (ANOVA, *p*=0.6).

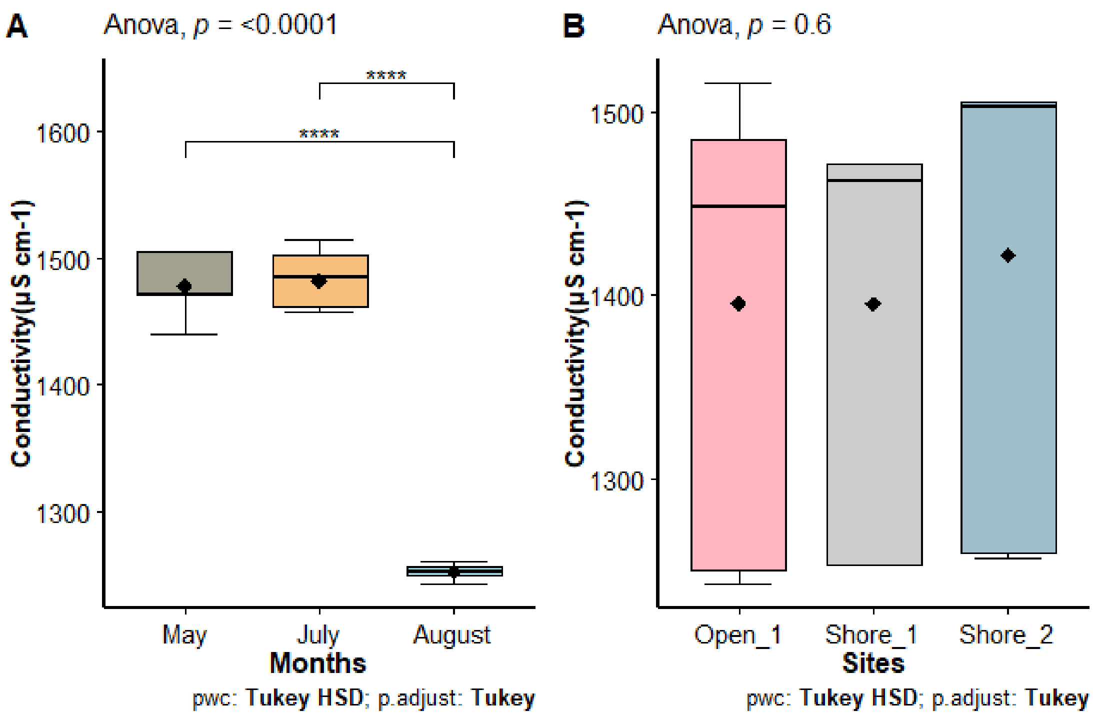

Figure 4.4: Box-and-whisker plot of EC (Electrical Conductivity) observed at the three sampling sites during the three sampling months. (A) Conductivity during Months (B) Conductivity among Sites.

TDS of the study lake was highest in May at Shore_2 (Mean =560±0mg/l) followed by that documented in July for Shore_2 (Mean =553.33±6.01mg/l). Nevertheless, the lowest value recorded was in August at Shore_1 (Mean =406.67±4.41mg/l) followed by that recorded in August at the Open water site (Mean =413.33±8.33mg/l). The TDS of the study lake showed marked temporal fluctuations (ANOVA; p <0.001) although there was no significant spatial variation (ANOVA, p =0.36).

TSS of LTA was the highest in July at Shore_1 (mean=351.33±32.54) followed by that observed in August at Shore_1 (mean=270.67±8.14). Nevertheless, the lowest value was recorded in May at Shore 2 (mean=158±15.37) followed by that observed in May at Shore 1 (194±11.14). The TSS of the study lake showed significant temporal and spatial fluctuations (ANOVA; p<0.001).

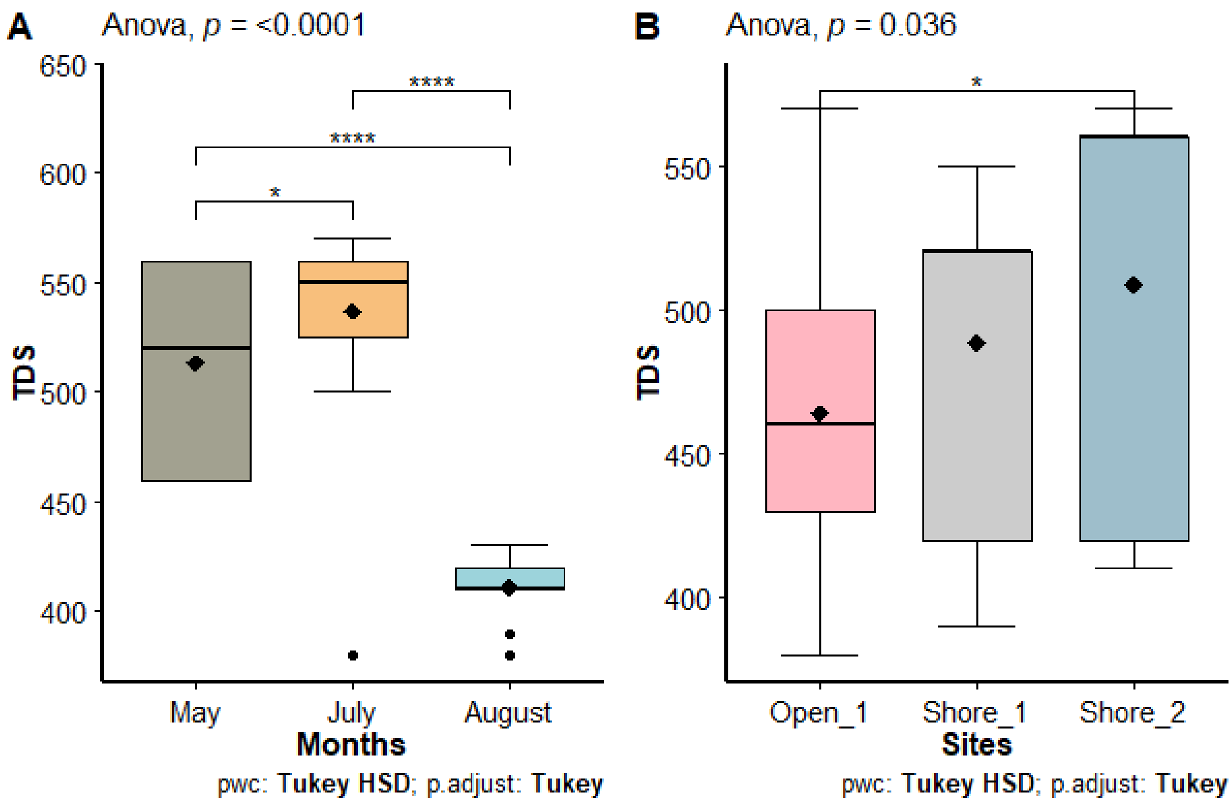

Figure 4.5: Box-and-whisker plot of TDS recorded at the three sampling sites during the three sampling Months. (A) TDS during Months (B) TDS among Sites.

The turbidity of the study lake was highest in July at the Open water site (Mean=766.67±0.44 NTU) followed by at Shore_1 (Mean =735.33±2.96 NTU). Nevertheless, the lowest turbidity was recorded in May at Shore_2 (Mean =350±2.57NTU) followed by Shore_1 (Mean =368.3±0.17NTU). The Turbidity of the study lake showed marked temporal fluctuations (ANOVA; p <0.001) although there was no significant spatial variation (ANOVA, p =0.28).

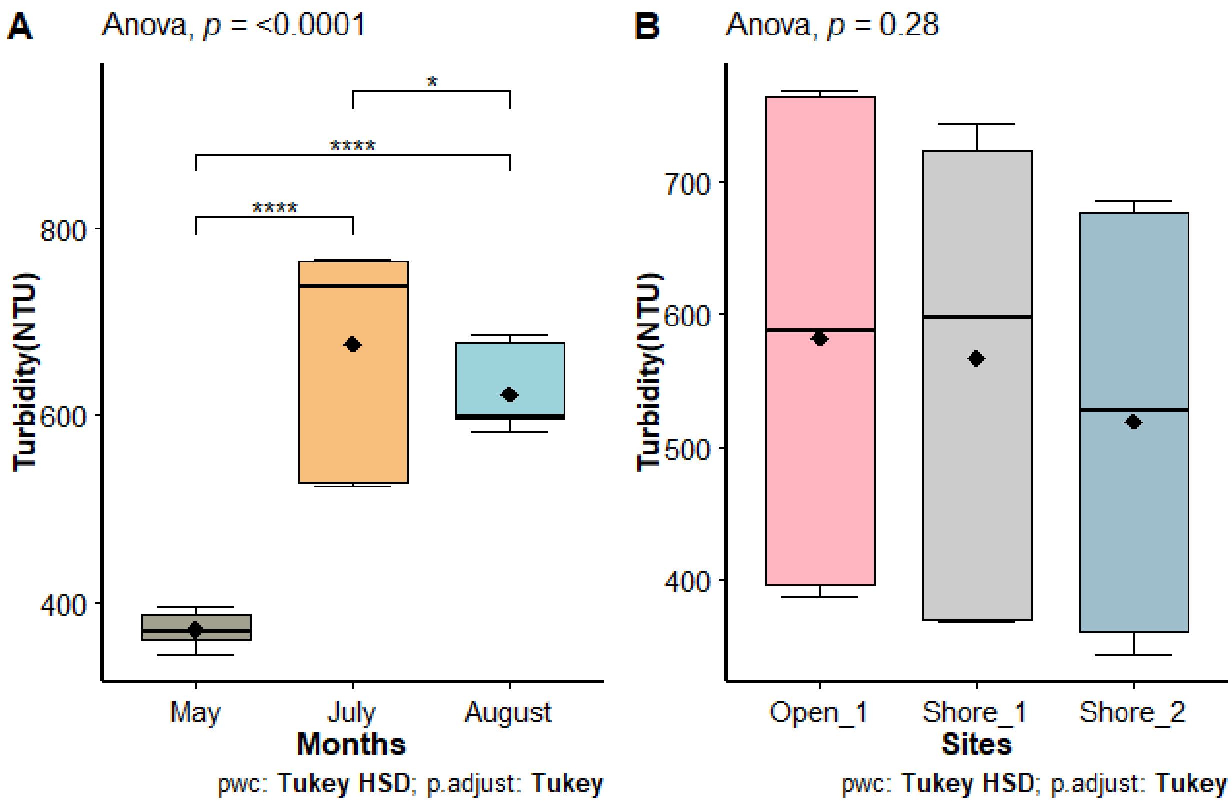

Figure 4.6: Box-and-whisker plot of Turbidity (NTU) recorded at the three sampling sites during the three sampling months. (A) Turbidity during Months (B) Turbidity among Sites.

Secchi Depth: In the mean value of secchi depth study lake were 0.095m, the euphotic depth (Zeu), the depth at which 1% of the surface photosynthetic active radiation (PAR) reaches with a mean value of 0. 225m.The Zeu is thin and primary productivity is light limited.

The trophic status of LTA, as determined by Carlson's Trophic State Index (TSI), indicates that the lake is in a eutrophic.

Inorganic Nutrients

The ammonia level in LTA was highest in August at Shore_2 (Mean=886.49±25.32 µg/L) followed by that observed in August at Shore_1 (Mean =802.52±26.63 µg/L). Nevertheless, the lowest level of ammonia was recorded in July at Shore_2 (Mean =349.35±7.65 µg/L) followed by that observed in July at Shore_1 (Mean =380.46.3±10.49). The ammonia concentrations in the study lake showed marked temporal fluctuations (ANOVA; p <0.001) although significant spatial variation was not observed (ANOVA, p =0.732).

The nitrate level in LTA was highest in July at the Open water site (Mean=519.35±13.97 µg/L) followed by that observed in July at Shore_1(Mean =492.88±15.29 µg/L). Nevertheless, the lowest nitrate level was recorded in May at the Open water site (Mean =244.48±3.84 µg/L) followed by that documented in May at the Shore_1 site (Mean =259.1±2.75). Nitrate concentration in the study lake showed marked temporal fluctuations (ANOVA; p <0.001) although there was no significant spatial variation (ANOVA, p =0.961).

Nitrite concentration in LTA was highest in July at Shore 1 (Mean=35.35±0.35 µg/L) followed by that recorded in July for Shore 2 (Mean =34.76±1.1 µg/L). Nevertheless, the lowest nitrate concentration was recorded in May at the Open water site (Mean =22.35±0.26µg/L) followed by that observed in May at Shore_1 (Mean =22.44±0.21 µg/L). Nitrite concentration in the study lake showed marked temporal fluctuations (ANOVA; p <0.001) and there was no significant spatial variation (ANOVA, p =0.815).

Total Phosphorus in LTA was highest in May at the Open site (Mean=482.04±14.01 µg/L) followed by that observed in May at Shore-2 (Mean =412±32.06µg/L). Nevertheless, the lowest Total phosphorous level was recorded in July at Shore-1 (Mean =150.69±5.01µg/L) followed by that documented in July at Shore_2 (Mean =170.91±11.84µg/L). Total Phosphorus of the study lake showed marked temporal fluctuations (ANOVA; p <0.001) and there was no significant spatial variation (ANOVA, p =0.456). The Redfield ratio of Total nitrogen to Total phosphorus (TN:TP) for the LTA is approximately 6.58:1.

Soluble reactive phosphate-phosphorus (SRP) in LTAwas highest in May at the Open water site (Mean=174.01±10.1 µg/L) followed by that observed in July at the Open site (Mean =162.67±5.2µg/L). Nevertheless, the lowest SRP level was recorded in July at Shore-1 (Mean =119±8.72 µg/L) followed by that documented in August at Shore-2 (Mean =125.72±2.48 µg/L). The SRP level of the study lake showed marked temporal and spatial fluctuations (ANOVA; p <0.001).

Silica concentration in LTAwas highest in May at Shore- 1 (Mean=154.7±7.85 µg/L) followed by that measured in May at Shore 2 (Mean =152±9.09µg/L). Nevertheless, the lowest silica level was recorded in July at Shore-1 (Mean =122.56±7.91µg/L) followed by that observed in July at Shore-2 (Mean =122.97±10.43 µg/L). Silica level of the study lake showed marked temporal fluctuations (ANOVA; p <0.001) and there was no significant spatial variation (ANOVA, p =0.113).

4.1.1 Principal Component Analysis

PCA can be a useful tool in the analysis of physiochemical parameters, as it can help identify important relationships and patterns in the data, as well as reduce the dimensionality of the data and simplify the interpretation of results.

The contribution of TDS, Temperature, nitrate (NO_3-N), nitrite (NO_2-N), ammonia ($NH_3^+NH_4^-$N), and Turbidity to water quality deterioration is high. TP, pH, and DO make moderate contribution, while SRP and SiO_2 are the least important in determining water quality.

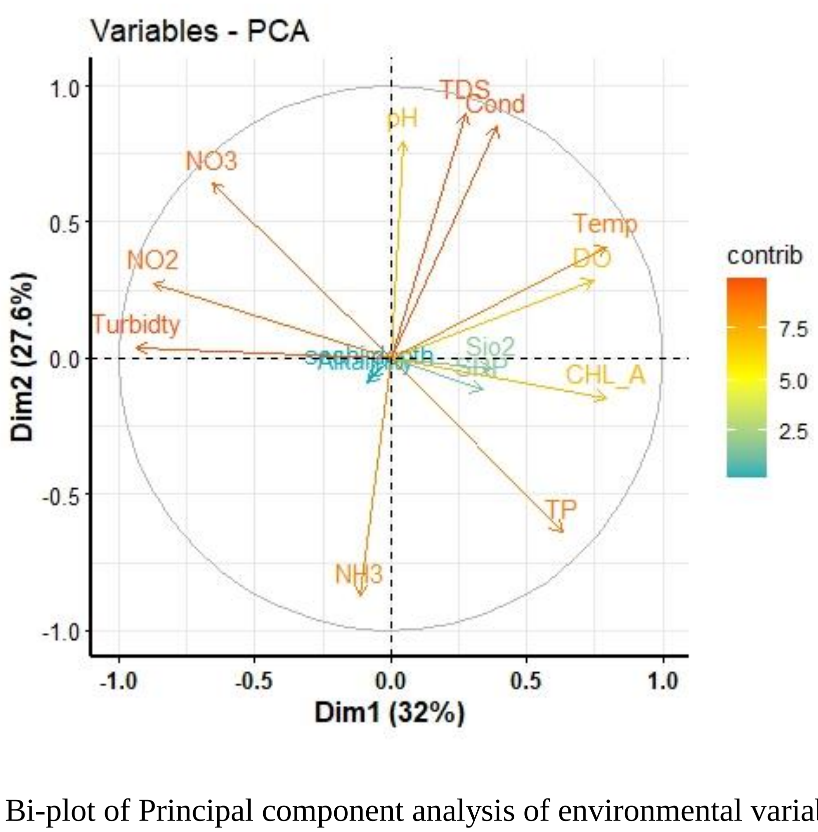

Figure 4.7: Bi-plot of Principal component analysis of environmental variables. The colors represent the extent of contribution: red = high contribution, yellow-moderate contribution and blue = low contribution.

4.2 Species composition, abundance and biomass of Phytoplankton

4.2.1 Species composition of Phytoplankton

32 phytoplankton taxa identified in samples collected from LTA during the present study (Table 3.1) belonged to 6 algal divisions: Bacillariophyta (Diatoms), Chlorophyta (Green algae), Cyanobacteria (Blue-green algae), Euglenophyta (Euglenoids), Cryptophyta (Cryptomonads), and Dinophyta (Dinoflagellates) Among the six algal divisions, Bacillariophyta was the most diverse (14 taxa, 54.5%) group of phytoplankton followed by Chlorophyta (9 taxa, 27.3%) and Cyanobacteria (7 taxa, 9.1%) while the flagellate groups, Euglenophyta (1 taxon, 3 %), Dinophyta (1 taxon, 3%), and Cryptophyta (1 taxon, 3%) were the most poorly represented in the phytoplankton community of the lake throughout the study period.

Table 4.1: Phytoplankton taxa recorded for Lake Tinshu Abaya

Division	**Species**	**Division**	**Species**
Bacillariophyta	*Asterionella* sp	Chlorophyta	*Ankistrodesmus bibraianus*
	Aulacoseira sp.		*Botryococcus* sp.
	Cyclotella sp.		*Chlorella* sp.
	Cymbella sp.		*Cosmarium contractum*
	Gomphonema sp.		*Oocystis* sp.
	Gyrosigma sp.		*Pediastrum duplex*
	Melosira sp		*Pediastrum boryanium*
	Navicula sp.		*Pediastrum simplex*
	Nitzchia sp.		*Scenedesmus* sp.
	Pinnularia sp.		
	Stauroneis sp.	Pyrrophyta	*Peridinium* sp.
	Surirella sp.		
	Synedria sp.		
	Thalassiosira sp.	Cryptophyta	*Cryptomonas* sp.
	Cylindrospoermopsis rasciborskii	Euglenophyta	*Euglena acus*
	Microcystis sp.		

Cyanobacteria	*Pseudoanabaena* sp.		
	Coelosphaerium kwuetzingianum		
	Oscillatoria sp.		
	Rhapidiopsis curvata		
	Cylindrospoermopsis curvispora		

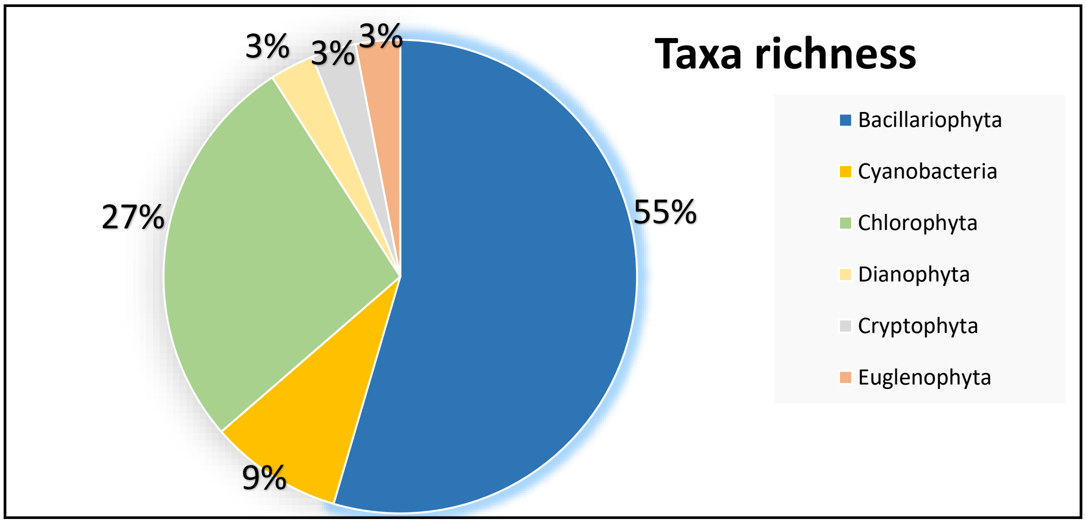

Figure 4.8: The percentage contribution of algal groups to the taxa richness of the phytoplankton community in LTA.

4.2.2 Phytoplankton Abundance in Lake Tinshu Abaya

A total of 12367 phytoplankton individuals belonging to 6 different division were identified. During the sampling period, total phytoplankton abundance in LTA ranged from 30 to 97098 cells/L The highest total phytoplankton abundance was found in May at the Open water site, while the lowest was recorded in July for Shore-1. The cyanobacteria were the most abundant during the three sampling months and followed by the Bacillariophyta. In May, higher number were recorded, 59.73 to 97098.21 cell/L at site Shore 2 Significantly ($P<0.001$) higher phytoplankton numerical abundance (97,098 cell/L) was recorded in May than during other sampling months (July and August), with total phytoplankton abundance at Shore-2 exceeding those of other sampling sites.

Table 4.2: Phytoplankton Abundance in Lake Tinshu Abaya

Month	Sites	Bacillariophyta	Chlorophyta	Cyanobacteria	Dinophyta	Cryptophyta	Euglenophyta
May	Open	7633.93	7165.18	90736.61	59.73	66.96	66.96
	Shore 1	4486.61	6294.64	88258.93	59.73	59.73	66.96
	Shore 2	6227.68	7834.82	97098.21	66.96	66.96	59.73
July	Open	4689.07	7116.15	70848.21	59.73	59.73	59.73
	Shore 1	4330.67	6003.20	64955.36	29.87	59.73	29.87
	Shore 2	5047.47	5794.13	65022.32	29.87	59.73	892.02
August	Open	6763.39	6160.71	55781.25	66.96	66.96	133.93
	Shore 1	6964.29	6763.39	58861.61	66.96	59.73	66.96
	Shore 2	7834.82	6696.43	59933.04	66.96	59.73	133.93

4.2.3 Biomass of phytoplankton

Phytoplankton biomass measured as chlorophyll-*a* concentration (Chl-a; µg L^{-1}) in Lake Tinshu Abaya ranged from 9.12 to 20.08 µg L^{-1}, averaging 16.47± 0.553 µg L^{-1} and with the highest biomass of phytoplankton occurring in May at all sampling sites, the lowest biomass of phytoplankton was recorded during the month of July at all sites, with a mean value of (11.68±0.5 µg L^{-1}). Biomass of phytoplankton differed significantly among the study months (ANOVA; $p<0.01$). It showed a trend of declining levels from May to July, which was followed by a small increase from July to August (rainy period) (Fig. 4.9).

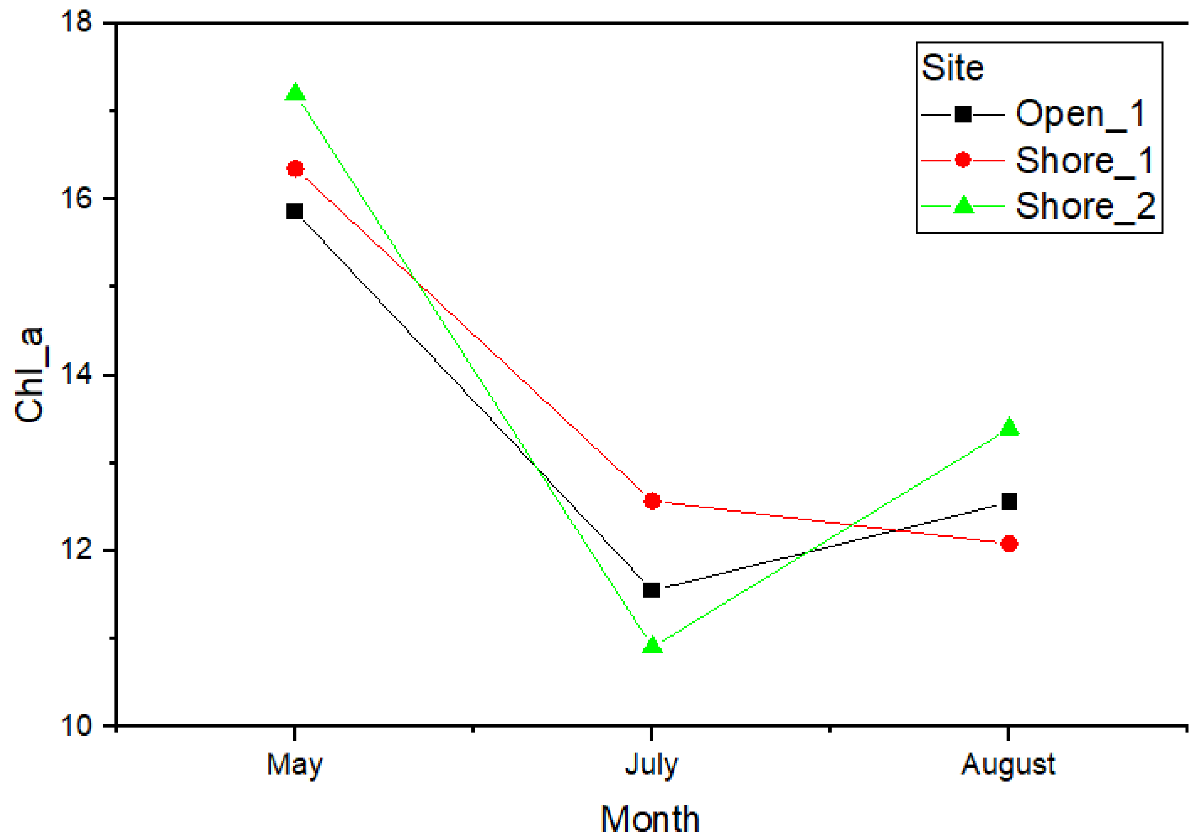

Figure 4.9: Spatio-temporal variations in phytoplankton biomass (as chlorophyll-a concentrations) in LTA at the Open water and shore sites of the present study period.

4.2.4 Phytoplankton abundance in relation to physico-chemical parameters

The correlation between the environmental parameters and distribution of dominant phytoplankton species like Botrococus sp., *Chlorella sp.*, *Cyclotella sp. Pediastrum duplex* and *Navicula* sp., *Microcystis* sp., *Cylindrospoermopsis curvispora and Cylindrospoermopsis rasciborskii* was analyzed using a constrained Redundancy Analyses (RDA) (Fig. 4.10). The first and second axes of the RDA biplot explained 65.03% of the cumulative percentage variance of species-environment relation of phytoplankton for the study area. *Botryococcus sp.*, *Chlorella* sp., *Cyclotella sp.*, *Pediastrum duplex* and *Navicula* sp. were correlated positively with inorganic nitrogen species (nitrite, nitrate and ammonia) and Turbidity, while *Microcystis* sp., *Cylindrospoermopsis curvispora and Cylindrospoermopsis rasciborskii* were correlated positively with TDS, conductivity and turbidity. *P. duplex* was correlated negatively with TDS, and conductivity and DO. *Botrococus brauni*, *Chlorella* sp., *Cyclotella sp.*, *Pediastrum duplex* and

Navicula sp. were correlated negatively and significantly with TDS, turbidity, temperature and conductivity. *C. curvispora* was strongly correlated with ammonia and nitrite.

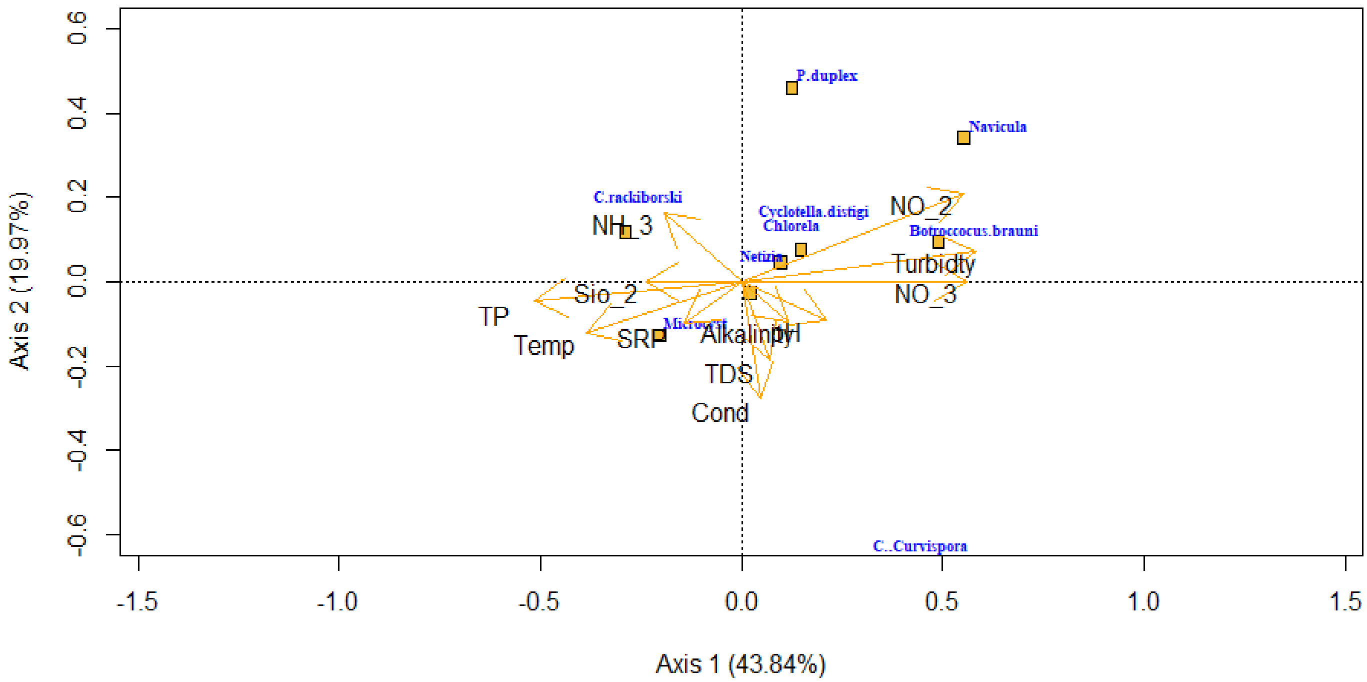

Figure 4.10: Tri-plot of the Redundancy Analyses (RDA) axis 1 explaining the variation 43.84 % and axis 2 explaining the variation 19.97%. dominant phytoplankton species (blue text) and environmental parameters (orange arrow).

4.3 Zooplankton composition, abundance and Biovolume

4.3.1 Zooplankton composition

In the present study, 25 taxa of zooplankton comprising large-sized micro crustaceans (copepods and cladocerans) and the small-bodied Rotifers were recorded in LTA during the study period.

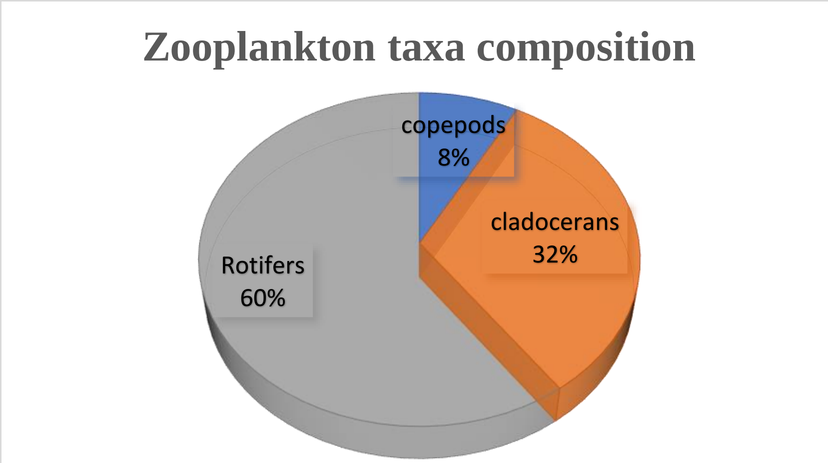

Figure 11: A pie chart showing the composition of zooplankton in LTA.

Table 4.3: Zooplankton taxa recorded for LTA

Copepods	**Cladocerans**	**Rotifers**
Thermocyclops decipiens	*Ceriodaphnia cornuta*	*Anuraeopsis fissa*
Mesocyclops aequatorialis	*Daphnia barbata*	*Ascomorpha ovalis*
	Daphnia cucullata	*Asplancha seieboldi*
	Daphnia pulex	*Brachionus angularis*
	Diaphanosoma brachyurum	*Brachionus calyciflorus*
	Diaphanosoma excisum	*Brachionus caudatus*
	Moina macrocopa	*Brachionus quadridentatus*
	Moina micrura	*Filinia longiseta*
		Filinia opoliensis
		Filinia pejleri
		Filinia terminalis
		Hexarthra intermedia
		Keratella tropica
		Lecane luna
		Trichocerca sp.

4.3.2 The Abundance of Zooplankton in LTA

Table 4.4: The abundance (individual L^{-1}) of zooplankton groups in Lake Tinshu Abaya at the open-water and both shore sites during the study period (May-August, 2022) (NOTE: SE is for standard error).

	Site	Copepods			Cladocerans			Rotifers		
		Min	Max	Mean± SE	Min	max	Mean± SE	Min	Max	Mean± SE
May	Open	78.7	442.1	260.41±181.70	5.7	24.1	13.95±5.4	9.71	534.22	136.58±100.36
	Shore 1	63.6	436.4	250.03±186.39	5.0	62.3	24.2±19.03	27.80	506.75	128.08±94.69
	Shore 2	85.1	439.1	262.08±177.01	5.0	52.6	20.9±15.79	21.70	551.63	132.23±104.8
July	Open	95.5	395.46	267.05±128.41	14.56	93.94	34.81±15.7	23.63	808.36	142.53±108.57
	Shore 1	51.2	351.24	262.33±88.91	6.03	88.92	30.86±15.08	21.60	763.65	137.02±102.32
	Shore 2	97.9	399.74	272.76±127.21	18.54	91.93	32.85±13.69	36.00	831.97	148.96±110.9
August	Open	81.1	240.6	210.88±29.75	13.95	44.12	33.45±6.26	9.11	725.68	147.95±96.9
	Shore 1	64.1	296.5	225.42±71.33	12.0	43.54	29.85±6.13	6.5	691.6	142.57±92.48
	Shore 2	31.3	277.8	213.6±82.31	16.96	62.82	46.25±9.29	12.05	719.68	153.41±95.64

There were no significant differences in among the three sites (ANOVA; $p > 0.05$), although estimates of total abundance of were higher for the open water than those of the shore sampling sites. The sampling months, however, varied significantly (ANOVA; $p < 0.001$). The three sites had peak overall abundance in July. Generally, during the rainy season (July and August), along with high water turbidity and high nutrient enrichment, the overall abundance of zooplanktons was relatively high. When compared to other months, May at LTA had relatively low relative zooplankton abundance, improving water clarity, low ambient inorganic nutrient levels, and high phytoplankton biomass.

4.3.3 The Biovolume of Zooplanktons in LTA

Zooplankton body length and width measurements were taken in random samples of rotifers, cladocerans and cyclopoid copepods (10 individuals of each species for each sampling site). For rotifers and copepods, the biovolume of the most abundant species was calculated. The biovolume of *Mesocyclops aequatorialis* averaged (mean=130.26+30.28μm^3/m^3), with values ranging from 33.42 μm^3/m3 to 321.12μm^3/m3). The biovolume of *Thermocyclops decipiens* averaged (mean=91.47+20.43 μm^3/m^3), with a range of (31.27μm^3/m^3- 148.43μm^3/m^3). Within the Rotifer group, the biovolume of *Brachionus calyciflorus* averaged (mean=33.60±6.5927μm^3/m^3 with estimated values varying between (11.72 and 58.61), while the biovolume of *Brachionus caudatus* averaged (mean=13.4427μm^3/m ±2.6227μm^3/m) ranging from 3.51m^3/m^3 to 20.933μm/m^3 *Keratella tropica*'s biovolume averaged 85.727mm^3/m^3 $\pm$ 3.827 mm^3/m^3, with values ranging from74.8 to 96.8 μm^3/m^3.

4.3.2 Physico-chemical variables and zooplankton abundance

Redundancy Analysis (RDA)

The correlation of some of the environmental parameters and distribution of dominant zooplankton species was analyzed using a constrained Redundancy Analyses (RDA) graph (Fig. 4). The first and second axes of RDA plot explained 79.9% of the cumulative percentage variance of species-environment relation of zooplankton for the study area. *T. decipiens* and *B. calyciflorus* were correlated positively with conductivity and TDS. *F. longstiea, Moina Micrura, B. caudatus* and *M. aequatorialis* were correlated positively with nitrite, nitrate and turbidity. *F. longstiea* and *Moina Micrura* were correlated negatively with conductivity and DO. *D. barbata* and *Keratella* *tropica* were correlated positively with DO and conductivity. *B. caudatus* and *M. aequatorialis* were correlated negatively and strongly with TDS, TSS, turbidity, temperature and conductivity. Napuli were strongly Positive correlated with ammonia and Turbidity.

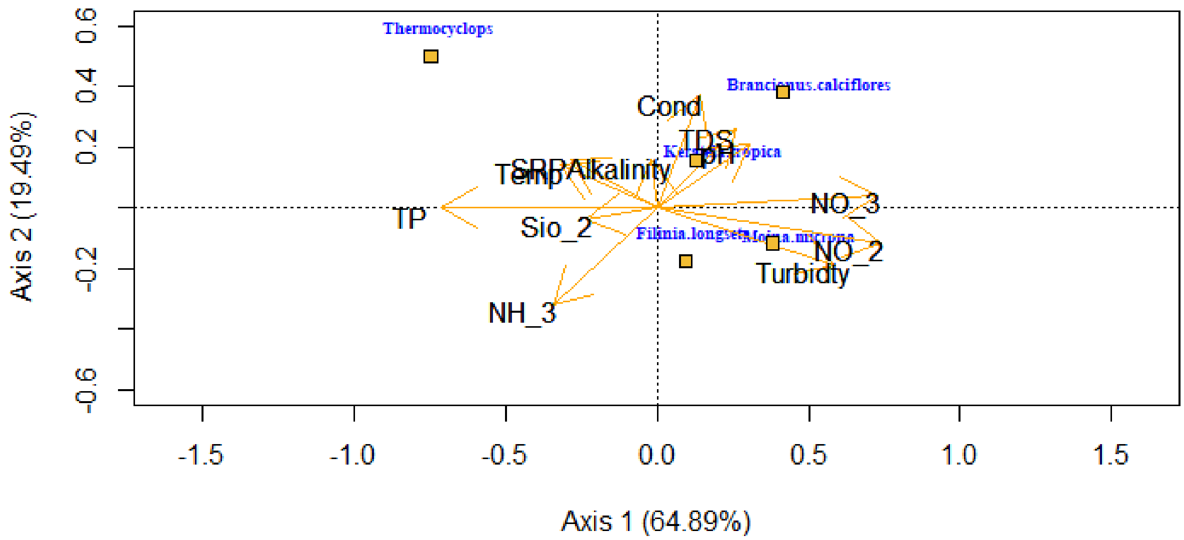

Figure 4.12: Tri-plot of the Redundancy Analyses (RDA) axis 1 explaining the variation 64.89 % and axis 2 explaining the variation 19.49%. Dominant phytoplankton species (blue text) and environmental parameters (orange arrow).

4.4 Trophic interaction and energy flow of Lake Tinshu Abaya

4.4.1The basic input data and biological parameters of groups calculated by Ecopath model

In this Ecopath model, the trophic level (TL) values of all ecological groups ranged from 1 to 3.56, with the primary producers (phytoplankton) and detritus at level 1; whereas the predatory birds were identified as the top predators, with a value of 3.56, followed by Nile Tilapia (2.79), Zoobenthos (2.35) etc. Similarly, ecotrophic efficiency (EE), values varied from 0 to 0.87. The top predators (birds) had a value 0, as no functional group consume on it. whereas the phytoplankton showed the highest ecotrophic efficiency value of 0.87. The higher ecotrophic efficiency indicates the higher utilization of the group within the ecosystem. The Production/consumption (P/Q) ratio value varies from 0.004 (birds) to 0.28 (herbivorous zooplankton).

Table4.5.

Table 4.5: The basic input data and biological parameters of groups as estimated by Ecopath for the ecosystem model

	Group name	**Trophic level**	**Biomass in habitat area(t/km²)**	**Production /biomass (/year)**	**Consumption /biomass (/year)**	**Ecotrophic Efficiency**	**Production / consumption /year)**
1	Birds	3.56129	0.022	0.25	58	0	0.00431
2	Nile tilapia (O. Niloticus)	2.79556	2.7	1.2	8.519	0.35517	0.14086
3	Zoobenthos	2.35556	2.43	16	84.64	0.60326	0.18904
4	Carnivorous Zooplankton	2.2	11.67	36	230	0.06541	0.15652
5	Herbivorous Zooplankton	2	5.11	270		0.409	0.28125
6	Phytoplankton	1	21.89	240		0.8715	
7	Detritus	1	1.2			0.75774	

4.4.2 Trophic structure

The trophic structure of Lake Tinshu Abaya has four trophic levels (Fig. 4.13). Birds were found to be the top predators of the system (Table 4.5). The trophic levels could be either fractional (Christensen and Pauly, 1992) or integers (Lindeman, 1942).

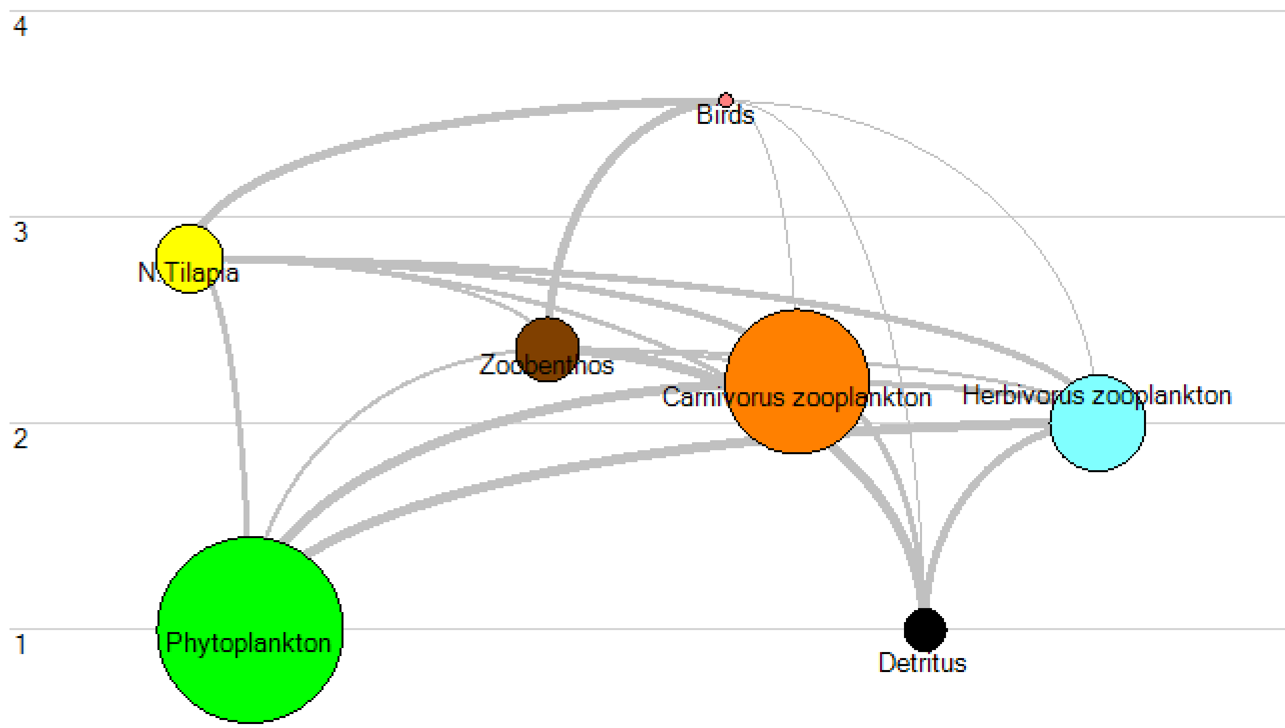

Figure 4.13: A diagram showing flows of energy in the mass balanced trophic model for LTA.

4.4.3 The omnivory indexes

The omnivory index for LTA is relatively high, 0.15. The omnivory index (OI) of each group in general is high, with most of the values greater than 0.2 (Table 4.6). It means the diet of several organisms is diversified and that these organisms can adapt to diet changes.

Table 4.6: The omnivory indexes in LTA

	Name	Omnivory index
1	Birds	0.07629
2	Nile Tilapia	0.28285
3	Zoo benthos	0.30133
4	Carnivorous zooplankton	0.16
5	Herbivorous zooplankton	0
6	Phytoplankton	0
7	Detritus	0.191

4.4.4 Keystone analysis

The keystone species were analyzed by three keystone indices KS1, KS2 KS3 and relative total impact. Mainly the higher trophic level organisms were identified as the keystone species.

Table 4.7: Keystone Species

	Group name	Keystone Index #1	Keystone Index#2	Relative total impact	Keystone Index#3
1	Birds	-0.475	2.825	0.484	0.303
2	Nile Tilapia	-0.544	0.694	0.441	0.0860
3	Zoobenthos	-0.338	0.943	0.703	0.386
4	Carnivores zooplankton	-0.294	0.415	1.000	0.141
5	Herbivores zooplankton	-0.412	0.575	0.634	0.119
6	Phytoplankton	-0.635	-0.0332	0.669	-0.335

4.4.6 Niche overlap

EwE produced niche overlap indices (predator overlap index, prey overlap index) based on the provided dietary input. In this study, LTA ecosystem, showed a high niche overlapping pattern among its component organisms

Table 4.8: Niche overlap Plot value

Niche overlap	
Predator: Prey	Predator: Prey
N. Tilapia: Zoobenthos	0.0281:0.502

N. Tilapia: Carnivorous zooplankton	0.00057:0.567
N. Tilapia: Herbivore zooplankton	0.0000237:0.412
Zoobenthos: Carnivorous zooplankton	0.971:0.476
Zoobenthos: Herbivore zooplankton	0.0393:0.652
Carnivorus zooplankton: Herbivore zooplankton	0.0396:0.9166

4.4.7 Trophic aggregation

Trophic levels were estimated by the program from the weighted average of prey trophic levels and varied from 1.0 for primary producers and detritus to 3.9 for top predators (piscivorous birds) in the artificial system. The trophic aggregation routine in Ecopath aggregated the groups with four effective trophic levels of Lake Tinshu Abaya. Primary producers (trophic level I) comprised the phytoplankton, and detritus.

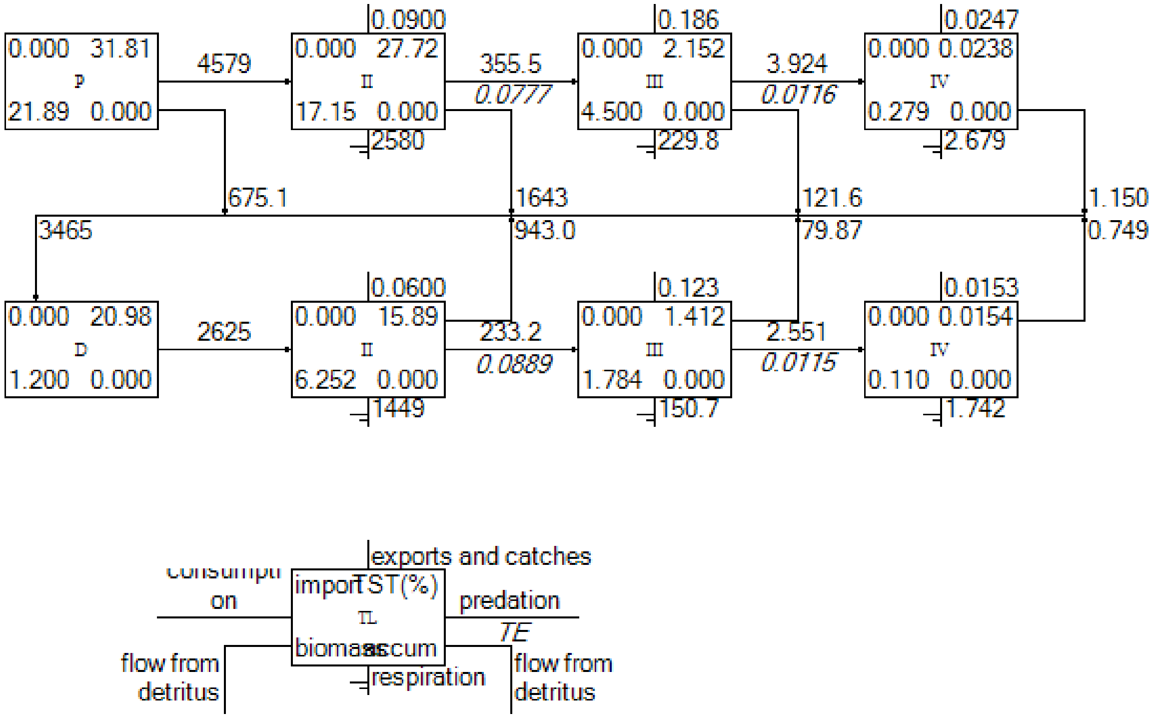

Figure 4.14: An aggregated flow diagram of the Lake Tinshu Abaya ecosystem organized by discrete trophic levels as the Lindeman Spine=Primary producers and D=detritus, representing both TL= I, TST =total system throughput=transfer efficiency and II – V represents trophic levels (TL).

4.4.8 Mixed trophic impact (MTI)

The MTI on LTA is illustrated in Fig. 4.15. It assesses the relative impact (direct or indirect) of increase of one group on others in the system. Prey and predator have positive and negative impact on each other, respectively. Almost all groups had negative impacts on themselves suggesting competition for same resources.

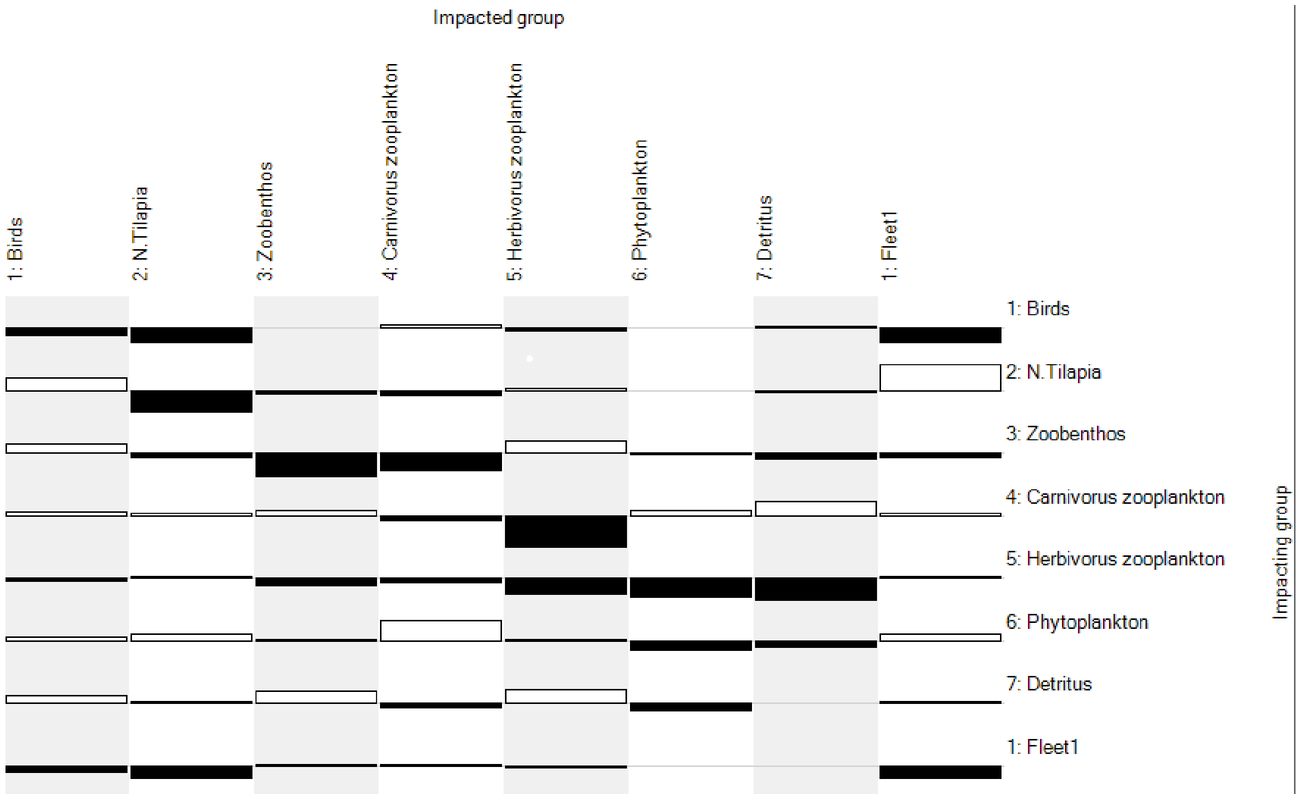

Figure 4.15: Ecopath, mixed trophic impact showing the impact that is both direct and indirect of an increase in the biomass of one functional group on others in the Lake environment. The bars represent relative effects and are comparable across groups. Positive impacts are presented above the baseline (white bars), whereas negative impacts are shown below (black bars).

4.4.9 Flows, transfer efficiency and system indices

Ecological indicators of this Ecopath model were total system throughput 10,668.79 t/km2 /year; Total primary production/total respiration 1.313; Ascendency 14218 flow bit; Overhead (system reserve) 42261; redundancy 16538; system robustness 0.86; and predator cycling Index 0.214. All the values of ecological indicators are shown in Table (4.8). These values indicate that this modified model is at developing state but the system is robust with better system reserve.

Table 4.8: Summary statistics for the artificial ecosystem model

Parameter	**Value**	**Units**
Sum of all consumption	7819.65	t/km^2/year
Sum of all exports	839.823	t/km^2/year
Sum of all respiratory flows	4413.78	t/km^2/year
Sum of all flows into detritus	3464.56	t/km^2/year
Total system throughput	16537.8	t/km^2/year
Sum of all production	7095.55	t/km^2/year
Mean trophic level of the catch	2.79556	
Gross efficiency (catch/net p.p.)	9.52E-05	
Calculated total net primary production	5253.6	t/km^2/year
Total primary production/total respiration	1.19027	
Net system production	839.823	t/km^2/year
Total primary production/total biomass	119.885	
Total biomass/total throughput	0.00265	t/km^2/year
Total biomass (excluding detritus)	43.822	t/km^2
Total catch	0.5	t/km^2/year
Connectance Index	0.57143	
System Omnivory Index	0.14965	
Ecopath pedigree	0.56522	
Measure of fit, t*	1.37032	
Shannon diversity index	1.28556	

5. Discussion

5.1 Physicochemical features

Temperature, DO and aggregate chemical parameters

Secchi depth is related to water clarity and is a measure of how deep light can penetrate into the water. In Lake Tinshu Abaya, shallow Secchi depths were observed. Large amounts of suspended soils or clay cause high turbidity thereby decreasing Secchi depths .The observed values show a decrease in euphotic depth by half when compared to those of the previous study(Yirga Enawgaw and Brook Lemma, 2018b).

The surface water temperature of LTA significantly differed among the sampling sites and sampling months ($p<0.001$). The decline in water temperature from its highest-level recorded May at both Shore sites to lower levels observed during the rainy sampling months (July and August) seems to have resulted from the decrease in the intensity of the sun and associated temperature of the air. LTA is categorized as a Polymictic lake because of its size and depth. The results of this study are consistent with that reported by Yirga Enawgaw and Brook Lemma (2018b) for the same sampling period.

According to Yirga Enawgaw and Brook Lemma (2018b) there is no significant variation in water temperature over the depth profile, which is also supported by the results of our study. The surface water temperature of LTA recorded during this study is comparable to those of other rift valley lakes of Ethiopia such as Lake Hawassa and Lake Ziway. The surface water temperature of LTA observed during this study is lower than those recorded for Lake Hawassa, which ranged from 20.56 to 28.30°C (Haile Melaku and Endale Tsegaye 2019), and Lake Ziway, which varied from 21 to 28°C(Dessie Tibebe *et al.*, 2022).

One of the most important parameters in a lake ecosystem is dissolved oxygen (DO) as extremely high or low levels can damage aquatic life and have an impact on water quality(Wetzel, 2001). The Dissolved oxygen of LTA showed significant temporal and spatial differences ($p<0.001$). The highest DO concentration was recorded during May at the Shore 1 and Shore 2 sampling sites, while the lowest DO level was observed during July at the Shore 1 and Shore 2. Despite this variance, the DO content in the lake supports aerobic life at those sites and months. The decaying organic matter from the allochthonous and high level of turbidity tend to decrease the level of DO during rainy seasons (Schmidt *et al.*, 2019). Oxygen depletion can cause significant changes in fish populations, resulting in mass fish kills in extreme cases(Greenbank, 1945). Dissolved Oxygen in the lakes is naturally generated by phytoplankton, algae and o aquatic plants as they photosynthesize(Rabaey *et al.*, 2021). It can also be mixed into a lake through aeration, like wind blowing across the water's surface. In shallow lakes, the amount of dissolved oxygen can vary greatly as the water of the lake is exposed to air that helps to mix with atmospheric air (Golosov *et al.*, 2012).

The pH level of LTA showed significant temporal differences ($p<0.001$) although there were no significant spatial variations ($p=0.5$). The highest pH was recorded during July at Shore_2 and Shore_1 after the flood of agricultural wastes and erosion from the mountain. The lowest pH was recorded during August at Shore_1. The present high values are slightly greater than those previously reported for the same lake (8.11-9.26)(Yirga Enawgaw and Brook Lemma, 2018b).This may have been caused by the significant changes in land use and the consequent increased soil erosion in the watershed, which increase the amount of chemicals entering the lake that influenced the pH levels. There is a substantial amount of total alkalinity in LTA, with the highest recorded value of (Mean $=6.67\pm0.21 meqL^{-1}$) at the open-water site and the lowest value of (Mean $=5.47\pm0.21 meqL^{-1}$) at the shore sites. The lakes's total alkalinity does not show significant spatial variation ($p > 0.05$), but there is temporal change ($p< 0.05$) that corresponds to the rainfall. When there is an excess of rain, total alkalinity declines because the concentration of total carbonates is diluted(Adane Sirage and Demeke Kifle, 2017).

Shore- 2 sampling site of LTA had the highest EC in May and July. The lowest Electrical Conductivity was, however, recorded at the Open and Shore-1 site in August. There were marked temporal changes in the electrical conductivity (ANOVA, $P <0.001$), although the spatial differences were not significant (ANOVA, $p=0.6$). The highest EC at the Shore-2 is attributable to the direct input through runoff from the agricultural lands, the grazing fields, and mountain erosion. Lake water conductivity can be influenced by a variety of factors such as watershed geology, watershed size in relation to lake size, wastewater from point sources, runoff from nonpoint sources, atmospheric inputs, evaporation rates, and some types of bacterial metabolism.(Pal *et al.*, 2015)

The present conductivity values are much higher than those previously reported for the same lake(Yirga Enawgaw and Brook Lemma, 2018b). The EC values of LTA are lower than those of Lake Hawassa that ranged from 741.7 to 1956.0 μS/cm(Haile Melaku and Endale Tsegaye 2019), although they are still higher than those Lake Ziway that ranges from 65.0 to 488 μS cm^{-1}(Dessie Tibebe *et al.*, 2022).

Total Dissolved Solids (TDS) is a measure of inorganic salts, organics, and other dissolved substances in water. The highest value of TDS of this study was twice that of the previous report(Yirga Enawgaw and Brook Lemma, 2018b). TDS levels can rise as a result of runoff from rivers and agricultural fields, as well as erosion from nearby mountains. The TDS levels of LTA in this study are lower than those of Lake Ziway, which ranged from 119.77 to 746.80 mg L^{-1} (Dessie Tibebe *et al.*, 2022) although they are still higher than those of Lake Hawassa that ranged from 420 to 455.6 mg L^{-1}(Begashaw Abate *et al.*, 2015). The concentration and composition of TDS in natural waters are influenced by runoff sediment, atmospheric precipitation, and hydrological regimes(Scannell and Jacobs, 2001).

The amount of light scattered by particles in water is measured as turbidity. The Turbidity of LTA was high throughout the sampling period. The Turbidity of the study lake showed marked temporal fluctuations (ANOVA; $P<0.001$) although the spatial differences were not statistically significant (ANOVA, $p=0.28$). The level of turbidity recorded in this study is four times that reported previously for the same lake by (Yirga Enawgaw and Brook Lemma, 2018b).

The reasons for this disparity may be that turbidity in lake water rises primarily as a result of soil erosion into the lake from the catchment. Deforestation in the watershed has been recorded, and the lake region is likely to have significant turbidity flows from tributaries transport suspended sediments into lakes (Davies-Colley and Nagels, 2008).Turbidity generally rises as the concentration of suspended sediments, algae, and organic matter rises(Gippel, 1995). Compared to other lakes, the turbidity of LTA is much higher than such lakes as Lake Hawassa whose turbidity ranged from 6.82 to 20.98 NTU.Thus,photosynthesis is limited by light (Begashaw Abate *et al.*, 2015).Lake Ziway mean value 163.5 NTU(Wondimu Tadiwos and Tenalem Ayenew, 2019) Lake Naivashia mean value of 199NTU(Ndungu, 2014)

Inorganic Nutrients

The ammonia of LTA was highest in May at Shore- 2 and Shore-1 the lowest ammonia recorded was in July at Shore-2 and in July at Shore-1. There was a relatively high concentration of ammonia during May, which may have resulted from reduced dilution due to the decrease in inflows during this month and high evaporation concentration. The subsequent lower level in July is attributable to the heavy rains that may have resulted in lake water dilution, the higher concentration of ammonia in August, during the month of relatively lower precipitation, may be associated with input through runoff from the agricultural fields over an extended period.

Nitrate is an important nutrient, but its presence in water resources promotes the growth of nuisance algae and macrophytes and causes eutrophication(Trevadi and Goel, 1986). The nitrate of LTA was highest in July and August and lowest in May. The increasing trend of nitrate concentration from May to August is probably due to increased input through runoff during rainy periods from the catchment area and is probably related to fertilizers applied on agricultural fields. Nitrite levels of LTA exhibited a temporal trend, which was broadly similar to that of nitrate. The levels of inorganic nitrogen of this study significantly differed in higher concentration from those of a previous reports (Yirga Enawgaw and Brook Lemma, 2018b) That could be due to high , human activities in recent period that have significantly increased the nitrogen inflow into water, including residential and agricultural sources, the quality of the water has decreased leading to nitrate overload (Kim *et al.*, 2019; Allan *et al.*, 2021). The concentrations of nitrate in LTA are lower than those of Lake Ziway that ranged from 0.1 to 5.26 mg L^{-1}(Dessie Tibebe *et al.*, 2022), while those of nitrite of LTA were higher than those of Lake Hawassa (mean value of 0.02mg/l).

Generally, the high levels of inorganic nitrogen family were recorded during the rainy season. High levels of nitrate in water can be a result of runoff or leakage from fertilized soil, wastewater, landfills, and animal feedlots (Jingsheng *et al.*, 2000).There were marked temporal fluctuations in the lake's inorganic nitrogen concentrations (ammonia, nitrite and nitrate), although their spatial variations were not significant ($p > 0.05$).The NH_3 value of LTA is lower than Lake Hawassa which ranges 3-1328 µg/l(Zinabu Gebre-Mariam, 2002).Also higher than the previous study 28.92-83.62 µg/l(Yirga Enawgaw and Brook Lemma, 2018b)

The highest total phosphorus concentrations were detected in LTA during the months of May and August, while, the lowest total phosphorous level was recorded in July. This may have resulted from the high rate of evaporation brought on by the high temperatures. Additionally, this month's high zooplankton abundance may have contributed to the rise in phosphorus concentration as when zooplankton consume phytoplankton, they assimilate the phosphorus into their bodies. When these zooplankton die or excrete waste, the phosphorus is released back into the water, contributing to the overall phosphorus concentration(Bai *et al.*, 2022).

The concentration of phosphorus increased during the month of August due to high levels of runoff that might have contained high levels of agricultural fertilizers. However, there seemed to be dilution of lake water with rainfall, which probably caused the low TP consentration observed in the month of July.The total phosphorus concentration of the study lake showed marked temporal fluctuations (ANOVA; $P<0.001$) although the spatial differences were not significant (ANOVA, $p=0.456$). Phosphorus can come from a variety of sources in the lake, including natural sources such as precipitation or geologic materials, as well as anthropogenic sources such as sewage, industrial discharges, wastewater treatment plant outputs, and fertilizers(Carpenter, 1981).A lake's trophic status can be determined by the concentration of phosphorus because it is typically the limiting nutrient for algal growth(Correll, 1999).The rate of phosphorus release increases as the water temperature rises, promoting its release(Holdren and Armstrong, 1980).The lake's phosphorus level rises as a result of higher zooplanktons becoming abundant (Gulati *et al.*, 1995). The value of TP is higher than the previous study(Yirga Enawgaw and Brook Lemma, 2018b).And also LTA higher than Lake Naivasha which is ranged from 200 to 420 µg/l (Obegi *et al.*, 2021).
The SRP (Soluble reactive phosphate-phosphorus) of the study lake showed marked temporal and spatial fluctuations (ANOVA; $P<0.001$). The results of this study's Redfield ratio (TN:TP) shows that total nitrogen is more readily available than phosphorus, therefore phosphorus becomes a limiting nuitrent. This finding indicated that the eutrophic nature of the lake is primarily due to non-algal turbidity, specifically sediment.

SRP concentrations were high during May, which may have resulted from evaporative concentration. The largest value coincided with the lowest water level in the lake, which may have led to the rise in the concentrations of nutrients due to evaporative concentration. The lake’s often fluctuating water level may also have had an effect on the concentration of SRP. During the dry season (May), the water level drops sharply and the lake recedes by many meters, stripping the shoreline of all vegetation and allowing the littoral sediments to dry. The value of SRP in Lake LTA is much higher than Lake Ziway 34µg/l, Lake Koka 16-81 µg/l, 9-37 µg/l (Eshete Assefa and Seyoum Mengistou, 2011).The value of lake SRP in LTA much lower than Lake Abijata 24_2790 µg/l(Zinabu Gebre-Mariam, 2002).

Silca of the study lake showed marked temporal fluctuations (ANOVA; $P<0.001$) even though there were no significant spatial differences (ANOVA, $p=0.113$). Silica (SiO_2) is an important component of the water chemistry in lakes, as it can influence the growth and abundance of diatoms, a type of phytoplankton, which form the base of the food chain in many marine and freshwater ecosystems. The silica content in a lake can be influenced by several factors, including input from surrounding watersheds, geological conditions, and water chemistry(Gibson *et al.*, 2000). The value of silica in this study have high difference from the previous study range value of 20.9-180.89µg/l(Yirga Enawgaw and Brook Lemma, 2018b).Lake Hawassa range value 19 -65 µg/l ,Lake Ziway 13.4-37.5 µg/l (Zinabu Gebre-Mariam, 2002)

5.2 Phytoplankton Composition, Abundance, and Biomass

Phytoplankton is important in any aquatic environment because it serves as a source of primary production, an indicator of pollution, and a view into how the ecosystem is functioning(Hubble and Harper, 2001).

In this study, the composition of identified algal species(32) is consistent with the reported earlier studies(37) for the same lake(Yirga Enawgaw and Brook Lemma, 2018a).

The difference in the total number of identified species is most likely due to the fact that the sampling period of this investigation was shorter than that of the previous study, and the occurrence of frequent top-down complete mixing, which might have resulted not to detect at least the rarer species. Because of the high non-algal turbidity (NAT), the species composition of LTA was high during the month of May and comparatively low during the month of July.

Compared to the nearby freshwater Rift Valley Lakes in Ethiopia, the species richness of phytoplankton of LTA is much lower than that of Lake Hawassa **(**100 phytoplankton species), (Elizabeth Kebede and Amha Belay, 1994)**,** Lake Ziway (67 species)(Elizabeth Kebede and Eva Willen, 1997). This could be related to the small surface area of LTA and very thin euphotic depth.

The Bacillariophyceae (Diatoms) contributed the greatest number of phytoplankton species in LTA, followed by the Chlorophyceae (green algae) and cyanophytes (Blue Green Algae). This result is in line with that of previous studies made on the same lake (Yirga Enawgaw and Brook Lemma, 2018a). Diatoms have been suggested to be particularly significant in lakes with high turbulence and turbidity due to their capacity to adapt to turbulent circumstances, efficient nutrient intake and relevance in food web dynamics (Huisman and Hulot, 2005).

Total phytoplankton abundance in LTA was highly dominated by the filamentous and colonial cyanophycean microalgae like M*icrocystis sp.* and *Cylindrospoermopsis raciborski*i followed by chlorophytes (Green algae) like *Cosmarium contractum* and *Chlorella* sp.

The dominance of cyanobacteria may be due to buoyancy regulation, tolerance to high temperature and pH, emission of allelopathic compounds (toxins), and resistance to grazing (Mowe *et al.*, 2015).

Cyanobacterial abundance such as M*icrocystis sp.* and *Cylindrospoermopsis raciborski*i in water is a massive concern. They might cause harm in a number of different ways, their blooms reduce the aesthetic appeal of recreational water, and decomposing algal blooms may result in anoxia due to high biological activity, killing fish and other aquatic life. They may put humans, birds, and aquatic wildlife in peril with their toxins(Richardson, 1997).

In the LTA, *Cylindrospoermopsis raciborski*i exhibited the highest abundance, particularly in May and August, surpassing other cyanobacteria phytoplankton species. This dominance of *Cylindrospermopsis raciborskii* in those months could potentially be associated with the lake's elevated turbidity levels and nutrient enrichment. The worldwide distribution and bloom formation of *C. raciborskii* are driven by its high plasticity and physiological tolerance to a wide range of environmental factors such as light, nutrients, and temperature, as well as antagonistic interactions with other phytoplankton species(Padisak, 1997; Beamud *et al.*, 2016). Additionally, the formation of allelochemicals by *C. raciborskii* has been reported as a benefit to this species' dominance by suppression of other cyanobacteria, such as *Microcystis sp.* (Mello *et al.*, 2012).

Seasonal variations may not have as much of an impact on the species composition of phytoplankton populations in tropical lakes and reservoirs as they do in temperate region. Instead nutrient availability, human activities, and biological interactions, can have an impact on the community structure and composition of phytoplankton in these ecosystems (Abbas, 2009; Figueredo and Giani, 2001).

Throughout the year, about equal proportions of every component, such as nutrient availability, water turbulence, grazing pressure, and so on, have an influence on phytoplankton abundance. The abundance of phytoplankton observed throughout the sampling months did not significantly differ between the sampling sites (ANOVA, $p<0.05$).

Phytoplankton biomass was low during July (rainy season) due to surface runoff carrying significant loads of sediments and suspended particles at the beginning of the rainy season, high turbidity blocks light (Adane Sirage and Demeke Kifle, 2017). In LTA, it was observed that the phytoplankton biomass was particularly high during the month of May. This increase in biomass can be attributed to the absence of rainfall during that period, allowing for ample sunlight availability, which promotes the growth of phytoplankton.

High non-algal turbidity (NAT) linked with either wind-induced resuspension processes or suspended soils that have been introduced from the surrounding landscape, might decrease overall phytoplankton biomass (Jones and Knowlton, 2005). By affecting light penetration and nutrient availability, wind-induced sediment resuspension can have an impact on planktonic primary production(Schallenberg and Burns, 2004).

As in LTA the majority of the phytoplankton biomass in L. Ziway and L. Awassa was composed of filamentous and chroococcal cyanobacteria, including various *Microcystis sp.*(Girma Tilahun and Gunnel Ahlgren, 2010).

The results of RDA showed that the majority of the green algae were positively associated temperature and the three main algal nutrients (ammonia, nitrate and silica). It shows that those parameters are important to phytoplankton, as has been shown in other studies(Demeke Kifle and Amha Belay, 1990).

5.3 Zooplankton Composition abundance and biovolume of zooplankton

5.3.1 Diversity and abundance of zooplankton

During the study period, 25 taxa of zooplankton were identified, including the small-bodied rotifers and the microcrustaceans copepods, and cladocerans with relatively large sizes. Rotifers made up the majority of the 25 species, contributing 60%, followed by cladocerans at 32% and copepods at 8%. The present finding is consistent with that of the previous study(Yirga Enawgaw and Brook Lemma, 2018). In Lake Hawassa, Rotifers contributed 69% of the total zooplankton richness, whereas cyclopoid copepods and cladocerans contributed 28% and 3%, respectively(Tadesse Fetahi and Seyoum Mengistou, 2014). In addition, the zooplankton community of Lake Ziway was composed of 81.13% Rotifers, 9.4% Cladocerans, and 5.6%

Copepods. (Adamneh Dagne *et al.*, 2008). Numerous scholars in various tropical aquatic ecosystems have also observed that rotifers are more diversified than microcrustaceans (e.g. Lake Tana, Lake Kuriftu and Lake Naivashia (Ayalew Wondie and Seyoum Mengistou, 2014; Eshete Assefa and Seyoum Mengistou, 2011; Harper *et al.*, 1990)

Zooplankton diversity is primarily impacted by the shoreline development index and the anthropogenic effect(Abd. Razak and Sharip, 2019). According to Tadesse Fetahi *et al.* (2011), environmental factors are the cause of differences in zooplankton's temporal and spatial patterns and zooplankton may also show different seasonality related to lake's seasonal mixing cycle (Seyoum Mengestou and Fernando, 1991).

Their great environmental tolerance and quick capacity to fill empty niches account for the enormous diversity of rotifers in different lakes (Phan *et al.*, 2021). Rotifers are essential to the system's energy flows in aquatic systems(Devetter and Sed'a, 2003). When rotifers and cladocerans share the same food resources, planktonic rotifers are frequently abundant and diversified when only small cladocerans are present, but they are rare when large cladocerans occur(Castro *et al.*, 2005; May and O'Hare, 2005).The same with the study finding in LTA that the rotifer are relatively highly abundant than cladocerans.

5.3.2 Abundance of Zooplankton

The analysis of zooplankton community abundance in LTA revealed that copepods were the most abundant component, followed by rotifers and cladocerans. One of the disparities between temperate and tropical lakes is the dominance of large cladocerans in temperate lakes as compared to that of smaller zooplankton species (such as rotifers) in tropical lakes. When compared to these two other zooplankton groups, adult survival in copepods is likely to be greater overall because they are capable of avoiding predators and copepods could also often survive for four days without food (Drenner, 1980; Hambright and Hall, 1992; Harfmann *et al.*, 2019)

Studies have shown that smaller zooplankton species benefit more compared to cladocerans in filamentous and colonial blue-green algae dominating system (Lürling, 2021). Given that cladocerans are filter feeders, consumption of such populations of algae slows down the rate of feed intake of the zooplankton due to toxicity, inadequate nourishment, and physical barriers to consumption (Fulton and Paer, 1988; Bouvy et al., 2000).

Determining the biovolume of zooplankton species helps in understanding their ecological significance, interactions with other organisms in the ecosystem, and potential influence on ecosystem dynamics and functioning. Zooplankton body length and breadth measurements were obtained from random samples of cladocerans, and cyclopoid copepods to calculate the biovolume of zooplanktons in samples from LTA. *Mesocyclops aequatorialis* had the greatest biovolume, followed by *Thermocyclops decipiens*. From rotifer group *Keratella tropica* had greatest biovolume followed by *Brachionus calciflorus.*

5.3.4 Physico-chemical variables and zooplankton abundance

Zooplankton is a diverse group of organisms that are essential to aquatic ecosystems because they provide food for fish and other higher aquatic animals. A number of physicochemical factors in their environment, such as water temperature, dissolved oxygen levels, pH, nutrition availability, and light intensity, have an impact on the development of zooplankton. Redundancy analysis of the relationship between physicochemical parameters and zooplankton species showed the degree to which physicochemical characteristics might affect changes in the density or composition of zooplankton species. *T. decipiens* and *B. calyciflorus* were correlated positively with conductivity and TDS. *T. decipiens* could be employed as a species that indicates eutrophic conditions and poor water quality, making it a crucial tool for the biomonitoring of aquatic ecosystems (Landa *et al.,* 2007). Rotifers are known for their ability to tolerate high levels of TDS, conductivity and salinity with low dissolved oxygen as compared to other groups(Waya and Chande, 2004).

Brachionus calyciflorus may live and flourish in water with extreme turbidity and high TDS levels. This tolerance may result from rotifers' capacity to osmoregulation and maintain homeostasis under difficult conditions(Wallace and Snell, 2010).

5.4 Trophic Interactions and Energy Flow

5.4.1 The basic input data and biological parameters of groups calculated by Ecopath for the artificial ecosystem model

Trophic interactions in aquatic ecosystems refer to feeding linkages between various organisms in the food chain. These interactions are crucial to the ecosystem's balance and health(Thompson *et al.*, 2012). Primary producers (such as phytoplankton), primary consumers (such as zooplankton), secondary consumers (such as fish), and tertiary consumers are the trophic levels in an aquatic environment. Each trophic level is energy and nutrient dependent on the one below it, with energy flowing from lower to higher trophic levels through consumption(Smith and Schindler, 2009).

Trophic interactions play a significant role in controlling population size and abundance in aquatic habitats. When a predator consumes a prey species, for example, it can control the population of the prey and keep it from overgrazing on primary producers. As a result, the ecosystem's nutritional and energy balance may be maintained. Moreover, trophic interactions can have an influence on the health and behavior of individual species, as well as the structure and function of the entire ecosystem. Understanding trophic interactions is therefore critical for managing and protecting aquatic ecosystems(Holomuzki *et al.*, 2010).

According to Christensen *et al.* (2005), ecotrophic efficiency is a key parameter in Ecopath models that indicates the efficiency of energy transfer between trophic levels. High ecotrophic efficiency values suggest that the ecosystem is functioning efficiently and energy is being efficiently transferred through the food web. The Ecotrophic efficiency (EE) of LTA was zero for the top predator (birds), while the phytoplankton showed the highest Ecotrophic efficiency of 0.87.

The high EE value for primary producers and detritus indicates a more efficient and productive ecosystem, where energy or biomass is effectively transferred to support higher trophic levels. The estimated EE value (< 1) for phytoplankton and detritus agrees with the observation of (Christensen and Pauly, 1992). The moderately higher EE of primary producer shows the ecosystem's potential bottom-up control(Traore *et al.*, 2008).

The gross efficiency (Catch/net p.p ratio) is 0.0000952. When the fisheries have a low gross efficiency, it means that the total ecological efficiency may be driven by efficient energy transfer across other trophic levels, excluding the fishery. The gross efficiency of LTA is much lower than Lake Awassa, 0.0014 (Tadesse Fetahi and Seyoum Mengistou, 2007), Lake Koka,0.08(Gashaw Tesfaye and Wolff, 2018) Lake Naivasha, 0.0009 and Lake Ihema, 0.0007 (Mavuti *et al.*, 1996).

Production/consumption (P/Q) values represent the flow of energy or biomass from one trophic level to another in a food web. The P/Q value is expressed as the ratio of the production of a given trophic level to the consumption of that trophic level by the level above it(Christensen and Pauly, 1992).

The Production/consumption (P/Q) values for LTA were low for birds (0.004) and high for herbivorous zooplankton (0.28). High P/Q values indicate that a given herbivore zooplankton was consuming a large amount of biomass or energy relative to its own production. This suggests that there is an inefficient transfer of energy between trophic levels. On the other hand, low P/Q values shows that a predatory bird is producing more biomass or energy than it is consuming. This may suggest that the trophic level is more efficient in transferring energy up the food web, or that there is a surplus of biomass available for consumption.

The P/R ratio is the ratio of total primary production (photosynthesis) to total respiration (the amount of organic matter consumed by organisms for energy). A ratio greater than one indicates that there is more production than consumption, whereas a ratio less than one indicates the reverse. In LTA P/R where greater one which is 1.190 which means there is more production than consumption. Immature ecosystems with high primary production have a high P/B ratio, whereas more mature systems with substantial biomass accumulation have a lower P/B ratio. This is because when an ecosystem grows, there is more biomass present, resulting in a reduced production-to-biomass ratio(Christensen, 1995).

The proportion of total primary production to total biomass (the total mass of living organisms in an ecosystem) in LTA were high with the value of 119.885 which explain early stages of ecosystem development, there is often a rapid growth of primary producers (phytoplankton) relative to the total biomass present in the system.

5.4.2 Keystone species

A keystone species is a species that has a disproportionately large impact on the structure and function of an ecosystem relative to its biomass or abundance(Paine, 1966) . These species are often not the most abundant or dominant species in an ecosystem, but their presence and activities can have a significant impact on the community composition and ecosystem dynamics(Estes *et al.*, 2011). In the current study, Carnivore zooplankton and Zoo benthos had a high keystone value of 1.00 and 0.703, respectively. According to a study by(Paine, 1966) keystone species are crucial in maintaining the balance and stability of ecosystems.

5.4.3 The omnivore index

The omnivores index in LTA ranges from 0 (zero) of phytoplankton and herbivorous zooplankton to that of Zoobenthos(0.301).According to Christensen *et al.* (2005), the consumer in question is specialized when the omnivory index value is 0, meaning it only consumes at one trophic level. A high value means the consumer consumes food from many trophic levels. There is no dimension to the omnivory index. So as the result shows, phytoplankton are primary producers, while herbivore zooplanktons are primary consumers that feed on only phytoplankton (i.e. they depend on a single trophic level). Zoo benthos has a high omnivore index and consumes food that is found at different trophic levels and also it has high feeding plasticity (flexibility and adaptability on feeding behavior).

5.4.4 Niche overlap

In Ecopath, niche overlap refers to the degree of resource utilization similarities between two or more species in an ecosystem. Niche overlap can have significant implications for community dynamics, such as resource competition and the possibility of species cohabitation or exclusion. The index is symmetrical and assumes values between 0 and 1.A value of 0 suggests that the two species do not share resources, 1 indicates complete overlap, and intermediate values show partial overlap in resource utilization(Pianka, 1974). In the current study, Zoo benthos and Carnivore zooplankton had high niche overlap that is due to their similar feeding habits and preferences. Zoobenthos are organisms that live on or near the bottom of aquatic environments, and they typically feed on detritus, algae, and other small organisms(Covich *et al.*, 1999).

Carnivorous zooplankton, on the other hand, are small aquatic animals that feed on other zooplankton. Both zoo benthos and carnivorous zooplankton occupy similar niches and co-exist within aquatic ecosystems, as they both rely on similar food sources and consume similar types of prey. As a result, they are likely to compete with each other for resources, leading to a high degree of niche overlap.

The least overlap of ecological niches is seen in Nile Tilapia (*Oreochromis niloticus*) and herbivorous zooplankton. Whereas herbivorous zooplankton eat phytoplankton and other aquatic microscopic organisms, Nile Tilapia is a freshwater fish that predominantly eats algae, aquatic plants, and debris (Mathewos Temesgen *et al.*, 2022). Nile tilapia are omnivorous, eating both plant materials and tiny invertebrates. They eat water plants, algae, debris, and even tiny zooplankton. Herbivorous zooplankton, on the other hand, eat mostly on phytoplankton, which consists of tiny algae. They have particular characteristics for filtering or feeding on these small plant cells. As a result, their niches do not overlap much, and they can coexist in the same ecosystem without much direct competition through resource partitioning. Furthermore, Nile Tilapia is a larger and more mobile species compared to herbivorous zooplankton, which are smaller and slow mover. This also reduces competition between the two species, as they have different abilities to access resources in various environment.

5.4.5 Mixed trophic impact

The Mixed Trophic Impact (MTI) method can be used to determine if changes in one group's biomass can have an impact on the biomass of other groups in the ecosystem(Valls *et al.*, 2015). All functional groups aside from detritus have a negative effect on themselves, which may lead to competition within the same resources. Most other groups benefited the most from phytoplankton and detritus. For instance, in LTA, phytoplankton has strong impact on herbivorous zooplankton and tilapia, which are their immediate grazers. Nile tilapia and herbivorous zooplankton, on the other hand, had a negative impact on phytoplankton. While predatory birds have a positive impact on Nile tilapia, zoo benthos has a strong negative effect on carnivorous and herbivorous zooplankton.

5.4.6 Flows, transfer efficiency and system indices

According to Pauly *et al.* (1998) a P/R ratio of 1 indicates that the production of organic matter in the ecosystem is balanced with the consumption of that organic matter through respiration. A P/R ratio greater than 1 suggests that production is greater than consumption, indicating a potentially healthy and productive ecosystem. Conversely, a P/R ratio less than 1 indicates that consumption is greater than production, suggesting that the ecosystem may be experiencing a loss of organic matter and potentially unsustainable.

In this study a P/R ratio of 1.19 was obtained and this suggests that the production of organic matter in the ecosystem is slightly higher than the consumption of that organic matter through respiration. This indicates a potentially healthy and productive ecosystem in which the organic matter is being utilized efficiently and that the ecosystem is able to support the current level of consumption and may have some potential for growth (Pauly *et al.*, 1998).

According to Christensen and Pauly (1993), the P/R ratio spans from 0.8 to 3.2 based on Ecopath model analysis, and LTA estimated value (P/R = 1.19) lies at the upper end of this range, indicating that LTA is an almost immature system with the low value of the total biomass/total throughput ratio (B/T). The low value of respiration in comparison to primary production, as well as the anticipated high value of Ascendency (25.17%), support this.

The total system throughput represents the size of the entire system in terms of flow(Ulanowicz, 1986). The total system throughput of LTA is T = 16537.81 t km^{-2} $year^{-1}$ (sum of consumption, exports, respiratory flows, and flows into detritus), showing that the system's size in terms of total flows appears to be rather small when compared to other tropical lakes in the country. LTA has a lower value than Lake Awassa, (T = 18,217 t km^{-2} $year^{-1}$)(Tadesse Fetahi and Seyoum Mengistou, 2007) and a higher value than Lake Koka (T = 9355 t km^{-2} $year^{-1}$)(Gashaw Tesfaye and Wolff, 2018).

The total transfer efficiency refers to the efficiency with which energy is transferred from one trophic level to the next, and it is calculated as the ratio of the production at one trophic level to the consumption at the previous level(Pauly *et al.*, 1998). The estimation of energy transfer efficiency between trophic levels in aquatic food webs is typically around 10% per trophic step(Pauly and Christensen, 1995). Energy decreases as it moves up the trophic levels because energy is lost as metabolic heat when the organisms from one trophic level are consumed by organisms from the next level(Brown *et al.*, 2004). The total transfer efficiency for LTA was 2.83%. The value of TE in LTA is lower those Lake Koka which is 12.0%(Gashaw Tesfaye and Wolff, 2018), Lake Hayq is 11.5% (Tadesse Fetahi *et al.*, 2011),Lake Tana is 6.2%(Ayalew Wondie *et al.*, 2012).

A Connectance index of 0.556 implies that the actual number of linkages in the food web is 55.6% of the maximum number of links possible. The connectance index of LTA higher than, Lake Hayq is 0.214(Tadesse Fetahi *et al.*, 2011), Lake Koka is (Gashaw Tesfaye and Wolff, 2018).Lake Volta is 0.43(Mensah *et al.*, 2019). This shows that the food web is moderately complex, with a reasonably high amount of connectedness between species. A higher Connectance index implies a more complex food web with more linkages between species, whereas a lower index indicates a simpler food network with fewer links(van Altena *et al.*, 2016; Ulanowicz, 1986).

The pedigree value of 56.5% shows that only 56.5% of the input data used to build the Ecopath model was based on actual data or direct observations; the remaining 43.5% was approximated or derived from other sources.

The ecosystem was more dependent on the pool of primary producers than on detritus to provide total system throughput, as evidenced by the fact that 98.5% of the system matter flow came from primary producers and just 1.41% from detritus.

The total catch of target species (0.5 t km^{-2}) seems to be very small compared to those of other tropical lakes including Lake Hawassa,(12.1 t km^{-2}, (Tadesse Fetahi and Seyoum Mengistou, 2007), Lake Hayq, (4.8 t km^{-2}:(Tadesse Fetahi *et al.*, 2011), and Lake Koka (3 t km^{-2}: (Gashaw Tesfaye and Wolff, 2018). There is no evidence of overfishing for the target species, even lower than the newly applied cautious E value (E = 0.4). This indicates potential for some more fisheries extension.

6.Conclusion and Recommendation

6.1 Conclusion

The results of the present study show that the lake is fresh, warm, turbidity and eutrophic. When compared to other lakes and reservoirs, the lake has more turbid water due to the significant sediment load from its watershed, where much of the plant cover has been removed. Nutrients are relatively high. However, the euphotic depth is very shallow and the lake is light limited. Redfield ratio (TN:TP) shows that total nitrogen is more readily available than phosphorus, therefore phosphorus becomes a limiting nuitrent. The eutrophic nature of the lake is primarily due to non-algal turbidity, specifically sediment.

The phytoplankton community of LTA exhibited low diversity, with filamentous and colonial cyanobacteria dominating the lake. Chlorophytes were in second place, followed by diatoms (Bacillariophyceae). The turbidity of the lake, which disadvantages other algae, might favor cyanobacteria to dominate in the lake. Phytoplankton biomass measured as chlorophyll-*a* concentration Chl-a; µg L-1) peaked in May, coincident with the high phytoplankton abundance.

The zooplankton community is moderately diverse, with more species of Rotifera. Copepods were the most abundant zooplankton in the lake, while the cladocerans were the least abundant. The biovolume of copepods was found to be greater than that of cladocerans and rotifers.

In this initial trophic modeling attempt for Lake Tinshu Abaya, it was possible to describe the lake ecosystem structure and it's the level of ecological maturity. The trophic levels of the groups varied from 1.00 for primary producers and detritus to 3.56 for predator birds. The pedigree value of shows that 56.5% of the input data used to build the Ecopath model was based on actual data or direct observations. The ecosystem maturity indices and Connectance index were found to be 1.19 and 0.556, respectively, indicating that the lake's system is at a developmental stage.

6.2 Recommendations

The establishment of a buffer zone could help to improve the lake's water quality by retaining nutrients entering the lake from the mountains and other catchment zones, thereby lowering the lake's high nitrogen load and turbidity. The growth of vegetative cover with a high capacity for absorbing nutrients, clay, and sediment, while creating little litter is the best strategy for promoting the lake's sustainability.

Implement strategies to reduce sediment intake into the lake. This might involve constructing sediment basins, implementing erosion control structures, and encouraging land management practices that limit sediment erosion and transport.

In light of the dominance of cyanobacteria (blue green algae) in the lake ecosystem, more research should be carried out to determine the possibility of cyanotoxin pollution and other factors that might result in effective management and conservation of the LTA.

LTA is utilized by the inhabitants for daily activities because it is the main source of water in the region fish can be a valuable source of nutrients, including protein and essential fatty acids, it is not the sole determinant of nutritional status. Since this study shows that the lake can support the growth of more fish, the local community may use it as a source of protein in their diet.

Considering the high abundance of copepods in the lake, it is imperative to conduct research on the reproductive biology and population dynamics of copepods. This investigation will facilitate a deeper comprehension of their life cycle, seasonal variations, and the factors that impact their abundance. Acquiring this knowledge is essential for effective management of copepod populations and accurate prediction of their responses to environmental changes.

We are able to describe the lake ecosystem and evaluate the level of ecological maturity in an ecosystem context with our initial attempt of trophic modeling for Lake Tinshu Abaya. It's vital to bear in mind that some of the input values were drawn from other systems, such as predatory birds, which might have given the model additional bias. To reduce bias from all sources and to quantify important input parameters, more thorough research must be done.

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Appendix i

Summary result of chemical future of the LTA

Variable	Months/Sites	May	July	August
NO3-N (µg/L)	Open_1	244.48±3.84	519.35±13.97	316.38±0.84
	Shore_1	259.1±2.57	492.88±15.29	320.58±4.38
	Shore_2	317.54±8.36	454.11±19.15	323.76±2.19
NO2-N (µg/L)	Open_1	22.35±0.26	33.73±0.16	33.09±0.34
	Shore_1	22.44±0.21	35.35±0.35	31.03±0.1
	Shore_2	23.82±0.27	34.76±1.1	38.05±0.15
NH3-N (µg/L)	Open_1	507.69±26.52	468.71±9.56	722.84±68.52
	Shore_1	478.87±12.23	380.46±10.49	802.52±26.63
	Shore_2	550.94±15.91	349.35±7.65	886.49±25.32
Tp (µg/L)	Open_1	482.04±14.01	195.8±14.52	370.0±37.68
	Shore_1	412±32.06	150.69±5.01	351.33±21.28
	Shore_2	418.0±27.65	170.91±11.84	371.78±17.2
SRP(µg/L)	Open_1	174.01±10.1	162.67±5.2	158.66±3.33
	Shore_1	150.62±8.83	119±8.72	146.44±4.83
	Shore_2	159.32±12.18	135.78±4.91	125.72±2.48
Sio2 (µg/L)	Open_1	129.88±11.41	122.97±10.43	131.82±11.57
	Shore_1	154.7±7.85	122.56±7.91	129.29±8.54
	Shore_2	152.55±9.09	131.73±5.09	149.16±8.8

Appendix ii

Major anthropogenic activities in and around Lake Tinshu Abaya

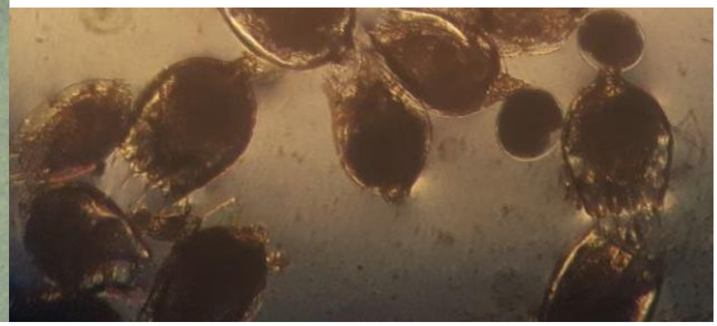
Using as drinking water sources for livestock

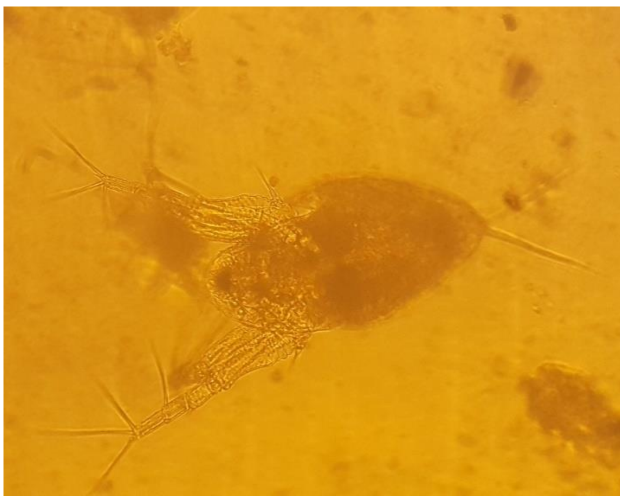
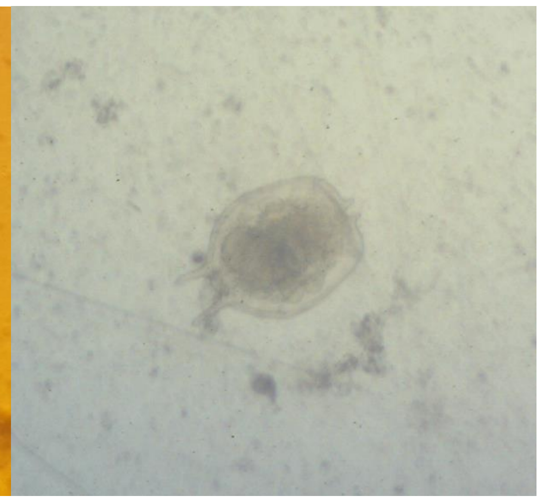
Peoples washing their mat at the shore

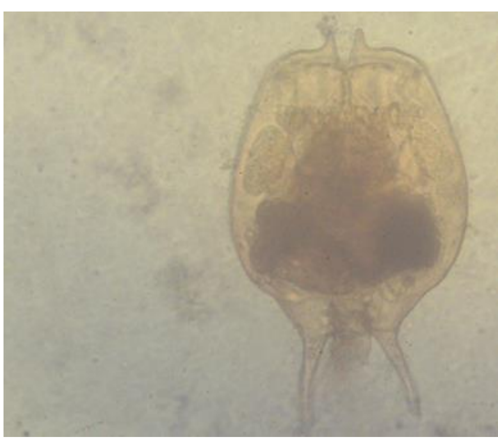
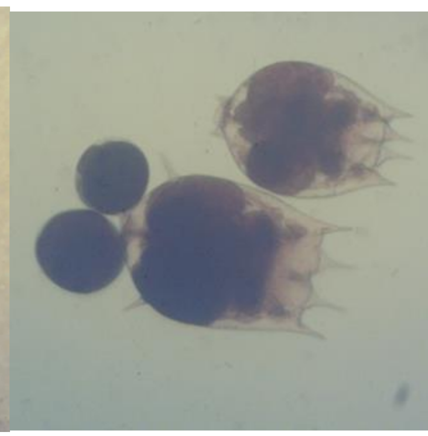
Agricultural activities around the lake catchment

Water abstraction from the lake for small scale irrigation

Appendix iii: -Some Zooplankton taxa recorded in LTA.


Appendix iv

Diet composition matrix, calculated from study data as a percentage of volume of prey groups, was constructed to create the ecosystem model (Christensen and Pauly, 1992).

	Prey	Predator				
		1	2	3	4	5
1	Birds					
2	Tilapia	0.51				

3	Zoobenthos	0.46	0.1	0.1		
4	Carnivorus zooplankton	0.01	0.3	0.1		
5	Herbivorus zooplankton	0.01	0.3	0.1	0.2	
6	Phytoplankton		0.18	0.1	0.6	0.6
7	Detritus	0.01	0.12	0.6	0.2	0.4
	Import	0	0	0	0	0
	Sum	1	1	1	1	1
	(1 - Sum)	0	0	0	0	0

Acceendancy

Source	Ascendency (flowbits)	Ascendency (%)	Overhead (flowbits)	Overhead (%)	Capacity (flowbits)	Capacity (%)
Import	0.000	0.000	0.000	0.000	0.000	0.000
Internal flow	7523	13.32	31787	56.28	39310	69.60
Export	1891	3.349	1726	3.055	3617	6.404
Respiration	4804	8.506	8748	15.49	13552	24.00
Total	14218	25.17	42261	74.83	56479	100.00

Mixed trophic plot

	Impacting / Impacted	Birds	N.Tilapia	Zoobenthos	Carnivorus zooplankton	Herbivorus zooplankton	Phytoplankton	Detritus	Fleet1
1	Birds	-0.158	-0.324	0.0150	0.0688	-0.0488	0.00740	0.0220	-0.324
2	N.Tilapia	0.271	-0.424	-0.0492	-0.106	0.0758	-0.0116	-0.0332	0.576
3	Zoobenthos	0.192	-0.0792	-0.490	-0.361	0.250	-0.0357	-0.137	-0.0792
4	Carnivorus zooplan...	0.0906	0.0602	0.119	-0.108	-0.663	0.112	0.308	0.0602
5	Herbivorus zooplan...	-0.0485	0.0393	-0.151	-0.0950	-0.331	-0.396	-0.473	0.0393
6	Phytoplankton	0.0931	0.156	0.0225	0.423	0.0417	-0.176	-0.119	0.156
7	Detritus	0.155	0.0462	0.263	-0.0881	0.293	-0.159	0.000	0.0462
8	Fleet1	-0.118	-0.250	0.0214	0.0463	-0.0329	0.00503	0.0144	-0.250

Predatory Overlap index

	Group name	1	2	3	4	5	6
1	Birds						
2	N.Tilapia		1.000				
3	Zoobenthos		0.0281	1.000			
4	Carnivorus zooplankton		0.000572	0.971	1.000		
5	Herbivorus zooplankton		0.000024	0.0393	0.0397	1.000	
6	Phytoplankton			0.00612	0.00619	0.464	1.000

Total Ascendancy

Source	Ascendency (flowbits)	Ascendency (%)	Overhead (flowbits)	Overhead (%)	Capacity (flowbits)	Capacity (%)
Import	0.000	0.000	0.000	0.000	0.000	0.000
Internal flow	7523	13.32	31787	56.28	39310	69.60
Export	1891	3.349	1726	3.055	3617	6.404
Respiration	4804	8.506	8748	15.49	13552	24.00
Total	14218	25.17	42261	74.83	56479	100.00

Transfer Efficiency

Source \ Trophic level	II	III	IV	V
Producer	7.767	1.156	2.428	2.173
Detritus	8.886	1.147	2.389	2.242
All flows	8.175	1.152	2.412	2.200
Proportion of total flow originating from detritus: 0.38				
Transfer efficiencies (calculated as geometric mean for TL II-IV)				
From primary producers: 2.793%				
From detritus: 2.898%				
Total: 2.833%				